Abstract

Populations of wild anadromous and resident salmonids continue to decline throughout much of the Pacific Northwest and northern California. Several stocks are presently listed as threatened or endangered under the Federal Endangered Species Act. Degradation of freshwater and estuarine habitats contribute substantially to this decline. Although Federal, State, and Tribal programs have been established, no coordinated, region-wide strategy exists to develop habitat conservation plans, foster habitat protection and restoration beyond minimum requirements on nonfederal lands, or encourage education and training.

This document provides the technical basis from which government agencies and landowners can develop and implement an ecosystem approach to habitat conservation planning, protection, and restoration of aquatic habitat on nonfederal lands. The report also describes a process for developing, approving, and monitoring habitat conservation plans, pre-listing agreements, and other conservation agreements for nonfederal lands to be consistent with the mandates of applicable legal requirements. Three parts constitute the body of the document. Chapters 1–10 supply the technical foundation for understanding salmonid conservation principles from an ecosystem perspective: over 50 years of reported scientific research has been synthesized to describe physical, chemical, and biological processes operating across the landscape, within riparian areas, and in aquatic ecosystems as well as the effects of human activities on these processes. Chapters 11–16 provide a general conceptual framework for achieving salmonid conservation on nonfederal lands in the Pacific Northwest, including specific guidelines for developing, monitoring, and implementing habitat conservation plans within the larger context of basin and regional conservation goals. An appendix lists information resources that landowners and agencies may find useful in developing and evaluating habitat conservation plans. Over 1100 sources are cited within this document.

The perspective we present in this document is anchored in the natural sciences. Although we touch on social, economic, and ethical concerns, an exhaustive discussion of these issues is beyond the report's scope. Nevertheless, our socioeconomic systems and values shape our perceptions of natural resources and drive our demands for them. The fate of salmonids in the Pacific Northwest is inextricably interwoven into this natural-cultural fabric. Just as conservation strategies that are not based on sound ecological principles will ultimately fail, ecological approaches that ignore socioeconomic values, political realities, and ethical issues are also at high risk of failure. In light of this inter-dependency between biological and social realms, we view this document as one piece of a conservation-restoration puzzle to be integrated into a more comprehensive assessment of what we as a society want and value, what legacy we wish leave to future generations, and how we can get there from here.

Key words

salmonids, aquatic ecosystems, aquatic habitat, land-use effects, environmental monitoring, environmental law, environmental regulations, disturbances, management systems, riparian habitat, watershed processes, habitat restoration, conservation

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Notice

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Preface

Populations of wild anadromous and resident salmonids are in decline throughout much of the Pacific Northwest and northern California. Several stocks are presently listed as threatened or endangered under the Federal Endangered Species Act (ESA), and continued losses are likely to result in additional ESA listings. A significant cause of salmonid declines is degradation of their freshwater and estuarine habitats. Although Federal, State, and Tribal conservation and restoration programs have been established, there is no coordinated, region-wide Federal strategy for developing habitat conservation plans pursuant to ESA, for fostering habitat protection and restoration beyond minimum ESA requirements on nonfederal lands, or for providing education and training in habitat protection and restoration strategies.

The National Marine Fisheries Service, the Environmental Protection Agency, and the Fish and Wildlife Service (the "Agencies") seek to develop 1) a training and outreach strategy to implement a coordinated ecosystem approach to ESA's habitat conservation planning as well as additional protection and restoration of aquatic habitat on nonfederal lands and 2) a process for developing, approving, and monitoring habitat conservation plans (HCPs), pre-listing agreements, and other conservation agreements for nonfederal lands that is consistent with the mandates of ESA, the Clean Water Act, and other applicable State and Federal requirements. This document provides the technical basis from which these goals can be accomplished. The primary intended audience is agency personnel who have background in the biological and physical sciences and who are responsible for overseeing land management activities. Use of technical terms that may be unfamiliar to some readers was at times unavoidable; consequently, the document may be less accessible to those without formal technical training in scientific disciplines.

The document is organized generally into three parts. Chapters 1–10 (Part I) provide the technical foundation for understanding salmonid conservation principles from an ecosystem perspective. We discuss the physical, chemical, and biological processes operating across the landscape, within riparian areas, and in aquatic ecosystems; these processes ultimately influence the ability of streams, rivers, and estuaries to support salmonids. Specific habitat requirements of salmonids during each life stage are detailed. We then review the effects of land-use practices on watershed processes and salmonid habitats, focusing on the impacts of logging, grazing, farming, mining, and urbanization on hydrology, sediment delivery, channel morphology, stream temperatures, and riparian function. An overview is presented on the importance of ocean variability in determining production of anadromous salmonids and the implications of this variability on restoration of freshwater habitats of salmonids. Next, land-use practices that minimize impacts to salmonids and their habitats are discussed, followed by a brief review of Federal laws that pertain to the conservation of salmonids on private lands. The Technical Foundation concludes with a review of strengths and weaknesses of existing programs for monitoring aquatic ecosystems; this chapter provides the basis for monitoring recommendations presented in Part II.

Chapters 11–16 (Part II) provide a general conceptual framework for achieving salmonid conservation on nonfederal lands in the Pacific Northwest, as well as specific guidelines for the development of Habitat Conservation Plans (HCPs) pursuant to the Endangered Species Act. We propose a hierarchical approach to the development and evaluation of HCPs and other conservation efforts, stressing the need for site- or watershed-level conservation efforts to be developed and evaluated within the larger context of basin and regional conservation goals. We outline critical issues that should be addressed at the scales of region and basin, watersheds, and individual sites while planning HCPs. We present details of specific elements for planning effective HCPs and criteria for evaluating the potential effectiveness of HCP provisions where such criteria are supported by current scientific information. Included in this discussion is an evaluation of the effectiveness of State rules for riparian management to protect specific processes that directly affect aquatic habitats. Compliance and assessment monitoring strategies for HCPs and other conservation efforts are proposed. The document concludes with a suggested strategy for implementing salmonid conservation efforts on nonfederal lands. An appendix (the third part) lists sources of data that landowners and agencies may find useful in developing and evaluating habitat conservation plans. Over 1100 sources are cited within this document and listed in the references section.

The perspective we present in this document found its anchor in the natural sciences. Although we touch on social, economic, and ethical concerns, an exhaustive discussion of these issues is beyond the scope of the document. Nevertheless, it is our socioeconomic systems and values that shape our perceptions of natural resources and drive our demands for them. The fate of salmonids in the Pacific Northwest is inextricably interwoven into this natural-cultural fabric. Just as conservation strategies that are not based on sound ecological principles will ultimately fail, ecological approaches that ignore socioeconomic values, political realities, and ethical issues are also at high risk of failure. Scientific information influences how society both views and values natural resources such as salmon. At the same time, social values influence where we devote our research efforts (and hence the strengths and weaknesses of our knowledge base) and the feasibility of implementing what is ecologically sound. In light of this interdependency between the biological and social realms, we view this document as one piece of a conservation and restoration puzzle to be integrated into a more comprehensive assessment of what we as a society want and value, what legacy we wish leave to future generations, and how we can get there from here.

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Finally, numerous others—too many to mention individually—have met with us to discus specific issues, have suggested documents to review or other experts to contact, have provided us materials to review, or have offered access to libraries and document collections.

Acronyms

AFS	American Fisheries Society
BIA	Bureau of Indian Affairs
BLM	Bureau of Land Management (DOI)
BOR	Bureau of Reclamation (DOI)
CDF	California Department of Forestry
CDFG	California Department of Fish & Game
CDWR	California Department of Water Resources
CFR	Code of Federal Regulations
CRFC	Columbia River Fisheries Council
CSWQCB	California State Water Quality Control Board
DO	dissolved oxygen
DOC	U.S. Department of Commerce
DOI	U.S. Department of Interior
EMAP	Environmental Monitoring and Assessment Program (EPA)
EPA	U.S. Environmental Protection Agency
ERS	Economic Research Service (USDA)
FEMAT	Forest Ecosystem Management Assessment Team
FS	Forest Service (USDA)
FWS	Fish and Wildlife Service (DOI)
GAO	General Accounting Office (U.S. Congress)
GIS	geographic information system
GPO	Government Printing Office
GS	Geologic Survey (DOI)
IDFG	Idaho Department of Fish & Game
IDWR	Idaho Department of Water Resources
LWD	large woody debris
MSG	California State Board of Forestry Monitoring Study Group
NIFC	Northwest Intertribal Fish Commission
NMFS	National Marine Fisheries Service (DOC, NOAA)
NOAA	National Oceanic and Atmospheric Administration (DOC)
NPCC	Northwest Power Planning Council
NPS	National Park Service (DOI)
NRC	National Research Council
NRCS	Natural Resources Conservation Service {formerly SCS} (USDA)
ODEQ	Oregon Department of Environmental Quality
ODF	Oregon Department of Forestry
ODFW	Oregon Department of Fish and Wildlife
ODSL	Oregon Division of State Lands
OSPRD	Oregon State Parks & Recreation Department
OWRRI	Oregon Water Resources Research Institute
PAH	polyaromatic hydrocarbon
PCB	polychlorinated biphenyl
PFMC	Pacific Fishery Management Council
PSMFC	Pacific States Marine Fisheries Commission
PSWQA	Puget Sound Water Quality Authority
PWD	Public Works Department, City of Olympia, Washington
REO	Regional Ecosystem Office [BLM, NMFS, FWS, BIA, EPA, FS]
SAB	Science Advisory Board {for EPA)
SCS	Soil Conservation Service (superseded by NRCS} (USDA)
TFW	Timber, Fish and Wildlife Group
TU	turbidity units
URL	universal resource locator
USC	U.S. Code
USDA	U.S. Department of Agriculture
WACT	Watershed Analysis Coordination Team
WDE	Washington Department of Ecology
WDF	Washington Department of Fisheries (superseded by WDFW)
WDFW	Washington Department of Fisheries and Wildlife {superseded WDF and WDW)
WDNR	Washington Department of Natural Resources
WDW	Washington Department of Wildlife (superseded by WDFW}
WFPB	Washington Forest Practices Board
WSSP	Washington State Shoreline Program

Parentheses () indicate the department to which an agency belongs.

Brackets [ ] indicate member agencies.

Squiggly brackets (} indicate additional information.

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1 Executive Summary: Part I

1.1 Introduction

As substantial evidence accumulates, concerns grow amid continuing declines of salmonids in the Pacific Northwest. Anadromous salmonids returning to the Columbia River to spawn have decreased from historical highs of 10–16 million wild fish to fewer than 2 million fish, mostly originating from hatcheries. At least 106 wild salmon stocks have been extirpated, 214 are at high or moderate risk of extinction, and many have been listed or are being reviewed for listing under the Endangered Species Act. Similarly, several resident species and stocks have also been proposed for listing. Salmon fisheries along coastal regions of Oregon and California have been dramatically curtailed because of dwindling numbers of fish and increasing concern for wild stocks. A number of natural and anthropogenic factors have contributed to these declines: hydropower operations, over exploitation, artificial propagation, climatic and oceanic changes, and destruction and degradation of habitat through land-use and water-use practices. Although the relative impact of these different factors varies among basins and river systems, habitat loss and degradation are considered contributing factors in the decline of most salmonid populations.

Part I of ***An** Ecosystem Approach to Salmonid Conservation* is intended to provide a comprehensive technical foundation for understanding salmonid conservation principles in an ecosystem context. Aquatic habitats critical to salmonids are the product of processes acting throughout watersheds and particularly within riparian areas along streams and rivers. This document depends on the premise that salmonid conservation can be achieved only by maintaining and restoring these processes and their natural rates. If ecosystems are allowed to function in a natural manner, habitat characteristics favorable to salmonids will result, and fish will be able to reinvade and populate historical habitats, recover from earlier stressors, and persist under natural disturbance regimes. This ecosystem-oriented approach complements recent Federal and State strategies that emphasize watershed and landscape-level functions of ecosystems for management and conservation of forest resources.

After briefly reviewing evidence of trends for Pacific Northwest salmonids (Chapter 2), we discuss physical, chemical, and biological processes that affect aquatic ecosystems and the salmonids that inhabit them (Chapters **3** and 4). Next, we present an overview of habitat requirements of salmonids, including elements that are essential to the general health of aquatic ecosystems, as well as specific habitat requirements at each life stage of salmonids (Chapter 5). We then discuss how human activities affect watershed and instream processes, focusing on effects of logging, grazing, agriculture (including irrigation withdrawal), mining, and urbanization (Chapter *6)***.** Effects of dams, species introductions (including hatchery practices), and salmon harvest are presented more briefly since these topics, while important in providing context for the document, were beyond the scope of this project. We also review the influence of climatic and oceanic conditions on salmonids and how these factors relate to salmonid conservation (Chapter 7). Next, we present an overview of management practices and programs that reduce the detrimental effects of human activities on salmonids (Chapter 8), followed by a discussion of Federal laws and regulations relevant to the conservation of salmonids (Chapter 9). Part I concludes with a review of strengths and weaknesses of existing monitoring programs for aquatic ecosystems (Chapter 10). In Part II of this document, we provide a general conceptual framework for achieving salmonid conservation on nonfederal lands in the Pacific Northwest as well as specific guidelines for the development of salmonid conservation plans, including Habitat Conservation Plans (HCPs), prepared pursuant to the Endangered Species Act. A separate executive summary (Chapter 11) describes major findings and recommendations related to conservation planning (Chapter 12–15).

This document focuses on anadromous salmonid species, including five Pacific salmon (chinook, coho, chum, pink, and sockeye), trout and char with both resident and anadromous forms (rainbow, cutthroat, and bull trout), and strictly resident species (mountain whitefish). The areal scope was limited to the portions of the States of California, Idaho, Oregon, and Washington that have supported salmonid populations. For many subject areas, we have relied heavily on comprehensive literature reviews and syntheses already available in the scientific literature. For subject areas where no such summaries were available, we have conducted more extensive literature reviews.

1.2 Physical and Chemical Processes

The physical and chemical characteristics of streams, rivers, lakes, and estuaries of the Pacific Northwest are the manifestation of processes operating at many temporal and spatial scales. Tectonic activity and glaciation have continually reshaped the landscape of the Pacific Northwest over millions of years. Alternating glacial and interglacial periods have caused changes in vegetative cover and geomorphic processes over significant portions of the region. Present climatic conditions have prevailed for the past 6,000 to 8,000 years, and modern coniferous forest communities developed over much of the coastal region within the last 2,000 to 5,000 years. In response to these changes, many river channels have shifted from unstable braided channels to relatively stable, meandering channels because the relative influence of hydrology, sediment delivery, and woody debris have changed.

Over periods of decades to centuries, large floods, fires, and mass wastings have been dominant natural disturbances influencing river channels. These disturbances can cause abrupt changes in habitat conditions, reconfiguring the stream channel, transporting streambed materials, depositing large quantities of coarse and fine sediments to streams, and altering hydrologic and nutrient cycling processes. These changes may persist for decades or more, affecting the relative suitability of habitats to various salmonids.

At the watershed and site levels, the major processes that affect the physical and chemical attributes of aquatic ecosystems are hydrology, sediment transport, heat energy transfer, nutrient cycling/solute transport, and delivery of large woody debris to streams. Runoff from the watershed affects stream habitats directly by determining the timing and quantity of streamflow, which control habitat availability and influence channel configuration, and indirectly by affecting the processes of energy transfer, sediment transfer, and nutrient cycling/solute transport. The amount of water reaching streams is a function of precipitation patterns, evapotranspiration losses, and infiltration rates, which in turn are affected by watershed characteristics including local climate, topography, soil type, slope, and vegetative cover. Hydrologic regimes of streams in the Pacific Northwest can be divided into three general patterns: rain-dominated systems, which are hydrologically flashy because of frequent rainstorms during the winter (coastal mountains, lowland valleys, and lower elevations of the Cascade and Sierra Nevada Mountains); transient-snow systems, which exhibit both rain and snow during the winter and may experience high flows associated with rain-on-snow events (mid-elevation of the Cascade, northern Sierra Nevada, and Olympic Mountains); and snow-dominated systems, where most precipitation falls as snow during the winter months and is delivered to streams in the spring as snow melts (higher elevations of the Cascade, Sierra Nevada, Olympic, and Rocky Mountains, and mid-elevation areas east of the Cascade/Sierra Crest).

Sediment from upland and riparian areas plays a major role in determining the nature and quality of salmonid habitats in streams, rivers, and estuaries. Sediment is generated from surface erosion and mass-wasting processes. Surface erosion occurs when soil particles are detached by wind, rain, overland flow, freeze-thaw, or other disturbance (animals, machinery) and transported to the stream channel. Mass wasting (slumps, earthflows, landslides, debris avalanches, and soil creep) results from weathering, freeze-thaw, soil saturation, groundwater flow, earthquakes, undercutting of streambanks, and wind stress transferred to soil by trees. Bank erosion and bedload movement occur naturally during high flows, but both may be exacerbated where riparian vegetation that stabilized banks is removed or when peak flows are increased by human activities. Watershed characteristics affecting sediment transport include climate, topography, geology, soil type and erodibility, vegetative cover, and riparian zone characteristics. West of the Cascades, mass wasting is the major source of sediments in undisturbed systems; east of the Cascades, both surface erosion and mass wasting may be important sources of sediments. In general, rain-dominated systems tend to yield more sediment that snow-dominated systems, although interbasin variability is high because of differences in topography, total precipitation, and soil type.

Stream temperatures influence virtually all aspects of salmonid biology and ecology, affecting the development, physiology, and behavior of fish, as well as mediating competitive, predator-prey, and disease-host relationships. Heat energy is transferred to streams and rivers by six processes: short-wave radiation (primarily solar), long-wave radiation, convective mixing with the air, evaporation, conduction with the stream bed, and advective mixing with inflow from ground water or tributaries. The temperature of streams represents a balancing of these factors. During the summer, incoming solar radiation is the dominant source of energy for smaller streams, though groundwater discharge may be locally important. Consequently, riparian vegetation plays a major role in controlling summer stream temperatures as may topographic features that provide shade. During the winter, direct solar radiation becomes less important because of lower sun angles, shorter days, and cloudier conditions. Stream characteristics, including width, depth, velocity, and substrate also determine the rate at which heat is gained or lost through radiation, convection, conduction, and evaporation. As streams become larger and less shaded downstream, the influence of both terrestrial vegetation and groundwater inputs diminishes, and temperatures tend to equilibrate with mean air temperatures.

Water is the primary agent dissolving and transporting solutes and particulate matter across the landscape, integrating processes of chemical delivery in precipitation, weathering, erosion, chemical exchange, physical adsorption and absorption, and biotic uptake and release. Climate, geology, and biological processes all influence the character and availability of inorganic solutes. The composition and age of parent rock determine the rate of weathering and hence the release of soluble materials. These dissolved materials are transported by surface and groundwater flow to streams. The biota of terrestrial, riparian, and aquatic ecosystems mediate the sources and cycling of major nutrients and associated organic solutes through processes such as photosynthesis, respiration, food uptake, migration, litter fall, and physical retention. Side channels on floodplains are areas of high nutrient uptake and processing because of low current velocities and extensive contact with the water column. Riparian vegetation may remove a significant proportion of the available phosphorous and nitrogen (60%–90%)and thus directly affects stream productivity.

Once in the stream, nutrients are transported downstream until they are taken up and processed by organisms and then released again, collectively termed "nutrient spiraling." The average distance over which one complete spiral occurs varies with stream characteristics, including retentive structures that physically trap particulate matter, stream size, water velocity, and the degree of contact between the water column and biological organisms inhabiting the stream bed. Simplification of channel structure increases nutrient spiral length, decreasing retention efficiency. Salmon and lamprey carcasses are also an integral part of nutrient cycling for both aquatic and riparian systems; thus declines in salmonids may cause more fundamental changes in ecosystem productivity than the simple loss of stocks or species.

Riparian and floodplain areas are the critical interface between terrestrial and aquatic ecosystems, serving to filter, retain, and process materials in transit from uplands to streams. Riparian vegetation plays a major role in providing shade to streams and overhanging cover used by salmonids. Streamside vegetation stabilizes stream banks by providing root mass to maintain bank integrity, by producing hydraulic roughness to slow water velocities, and by promoting bank building through retention of sediments. Riparian vegetation also provides much of the organic litter required to support biotic activity within the stream as well as the large woody debris needed to create physical structure, develop pool-riffle characteristics, retain gravels and organic litter, provide substrate for aquatic invertebrates, moderate flood disturbances, and provide refugia for organisms during floods. Large woody debris performs important functions in streams, increasing channel complexity, creating hydraulic heterogeneity, and providing cover for fish. Large wood also provides critical habitat heterogeneity and cover in lakes, estuaries, and the ocean. In addition to the aquatic functions that riparian areas perform, they typically provide habitat and create unique microclimates important to a majority of the wildlife occupying the watershed.

1.3 Biological Processes

The physiology and behavior of organisms, the dynamics and evolution of populations, and the trophic structure of aquatic communities are influenced by the spatial and temporal patterns of water quantity and velocity, temperature, substrate, physical structure, and dissolved materials. At the organism level, survival of salmonids depends on their ability to carry out basic biological and physiological functions including feeding, growth, respiration, smoltification, migration, and reproduction. All of the habitat characteristics listed above influence the quality and amount of food energy available, the amount of energy expended for metabolic processes, and hence the amount available for growth, migration, and reproduction.

Each phase of the salmonid life cycle—adult maturation and migration, spawning, incubation of embryos and alevins, emergence of fry, juvenile rearing, and smolt migration—may require utilization of and access to distinct habitats. The strong homing ability of salmonids has led to the formation of numerous, relatively isolated stocks, each adapted to the specific environmental conditions found in its natal and rearing habitats. This adaptation is reflected in the wide diversity of life histories exhibited by the salmonids of the Pacific Northwest. A major concern is that land use and water use have reduced habitat diversity through loss or simplification of habitat, which in turn has reduced the life-history diversity exhibited in the salmonid populations. At larger spatial scales, groups of populations or "metapopulations" interact infrequently through straying or dispersal. Metapopulation theory suggests local populations within metapopulations periodically go extinct and are recolonized and that metapopulations will persist if recolonization rates exceed extinction rates. The core-satellite metapopulation model proposes that extinction probability is not equal among populations and that certain extinction-resistant populations are important "seed" sources of recolonizers for habitats made vacant by extinction. Conservation of salmonids thus depends on maintaining: connectivity among habitats to allow reinvasion of vacant habitats, sufficient genetic diversity to allow successful recolonization of these habitats, and refugia from which dispersal can occur. The concept of Evolutionarily Significant Units presently being used by Federal agencies to determine appropriate units of conservation for salmonids is based in part on these metapopulation considerations.

Biotic communities in aquatic ecosystems are influenced by predator-prey, competitive, and disease- or parasite-host relationships within and among species. Current theory suggests that disturbance plays a major role in influencing the outcome of these interactions and, thus, in determining community or assemblage structure. Two models appear applicable to stream communities. The "intermediate disturbance hypothesis" argues that diversity is greatest in systems experiencing intermediate disturbance, because neither colonizers (favored by frequent disturbance) nor superior competitors (favored by infrequent disturbance) can maintain dominance. The "dynamic equilibrium model" proposes that community structure is a function of growth rates, rates of competitive exclusion, and frequency of population reductions; inferior competitors persist if disturbance occurs often enough to prevent competitive exclusion, but species with long life cycles are lost if disturbance is too frequent. Both of these theories suggest that increases in disturbance frequency caused by human activities are likely to alter community structure.

Food webs in aquatic systems are highly complex, consisting of many species representing several trophic levels. These food webs can be highly modified by environmental changes in light energy or nutrient inputs; alterations of streamflow, temperature, or substrate; and introductions of non-native organisms. Changes in physical habitat characteristics can alter competitive interactions within and among species. Similarly, changes in temperature or flow regimes may favor species that prey on salmonids, such as northern squawfish and a host of introduced predators, Salmonids are affected by a variety of bacterial, viral, fungal, and microparasitic pathogens. Both the immune system of fishes and the virulence of pathogens are greatly affected by environmental conditions, especially temperature; thus, alteration of temperature, substrate, and flow may increase the incidence of epizootics.

1.4 Salmonid Habitat Requirements

Operating throughout the watershed and across the landscape, all of the physical, chemical, and biological processes discussed above affect the features and characteristics of aquatic habitats from headwater streams and lakes to estuaries and the ocean. To protect or restore desirable habitat requires that the natural processes producing those features and characteristics must be maintained or restored. Four general principles should be considered when determining habitat requirements of salmonids:

- Watersheds and streams differ in their flow, temperature, sedimentation, nutrients, physical structure, and biological components.
- Fish populations adapt and have adapted—biochemically, physiologically, morphologically, and behaviorally—to the natural environmental fluctuations that they experience and to the biota with which they share the stream, lake, or estuary.
- Specific habitat requirements of salmonids differ among species and life-history types, and these requirements change with season, life stage, and the presence of other biota.
- Aquatic ecosystems change over evolutionary time.

Consequently, there are no simple definitions of salmonid habitat requirements, and the goal of salmonid conservation should be to maintain habitat elements within the natural range for the particular system.

Five general classes of features or characteristics determine the suitability of aquatic habitats for salmonids: flow regime, water quality, habitat structure, food (energy) source, and biotic interactions. Flow regimes directly influence the depth and velocity of water and the total available habitat space for salmonids and their food organisms as well as perform other functions such as redistributing sediments, flushing gravels, and dispersing vegetation propagules. Water quality requirements include cool temperatures, high dissolved oxygen, natural nutrient concentrations, and low levels of pollutants. Salmonids prefer cold water, and temperatures above 25°C are lethal to most species; individual species have specific preference ranges that vary by life stage. Variation in temperature is required to trigger spawning, support growth, initiate smoltification, and enable other parts of the salmonid life cycle. Salmonids require well oxygenated water (> 6 mg/l) throughout their life cycles, and any level below saturation can be detrimental. Nutrient levels vary among streams and must be sufficient to support natural plant and animal assemblages, Important structural attributes of streams include pools, riffles, substrate, cover (e.g., undercut banks, overhanging vegetation), depth, and hydraulic complexity. The presence of large woody debris enhances channel complexity, creating hydraulic heterogeneity, pools, side channels, back eddies, and other features that are used by salmonids and other aquatic organisms. Maintaining adequate food sources depends upon maintaining natural inputs of allochthonous material (type, amount, and timing) as well as physical structures needed to retain these materials. Normal biotic interactions also must be maintained to ensure the health of aquatic ecosystems, including competitive, predator-prey, and disease-parasite relations.

Stream habitat and channel features vary markedly from headwater streams to the estuaries and ocean. Salmonids, particularly anadromous species, use the entire range of habitats encountered during completion of their life cycles. The diversity of life histories exhibited by salmonids has developed to accommodate and fully exploit the range of habitats encountered. Loss of specific elements of habitat diversity may reduce the diversity exhibited in the salmonids' life histories, which in turn may influence the ability of these fish to adapt to natural and anthropogenic change.

Habitat requirements vary by life stage. During spawning migrations, adult salmon require water of high quality (cool temperatures or thermal refugia, dissolved oxygen near 100%, and low turbidity); adequate flows and depths to allow passage over barriers to reach spawning sites; and sufficient holding and resting sites. Spawning areas are selected on the basis of species-specific requirements of flow, water quality, substrate size, and groundwater upwelling. Embryo survival and fry emergence depend upon substrate conditions, including gravel size, porosity, permeability, and oxygen levels; substrate stability during high flows; and appropriate water temperatures (< 14°C for most species, but < 6°C for bull trout). Habitat requirements for rearing juveniles of anadromous species and adults of resident species also vary with species and size. Microhabitat requirements for holding, feeding, and resting each differ, and these requirements change with season. Migration of juveniles to rearing areas (whether the ocean, lakes, or other stream reaches) requires unobstructed access to these habitats. Physical, chemical, and thermal conditions may all impede migrations of juvenile fish.

1.5 Effects of Human Activities on Watershed Processes, Salmonids, and Their Habitats

Land-use practices, including forestry, grazing, agriculture, urbanization, and mining can substantially alter watershed processes, resulting in degradation of streams, lakes, and estuaries. Logging and grazing affect the greatest percentage of lands in the Pacific Northwest, but effects of agriculture, urbanization, and mining may result in a higher degree of local disturbance. Most of the alterations from land-use practices in upland areas result from changes in vegetation and soil characteristics, which in turn affect the quantity and routing of water, sediments, nutrients, and other dissolved materials delivered to streams. In addition, application of chemical fertilizers and biocides can affect water quality. Activities within the riparian zone can alter shading (and hence stream temperature), transport and supply of sediment, inputs of organic litter and large wood, bank stability, seasonal streamflow regimes, and flood dynamics. Dams, irrigation diversions, and road crossings hinder migrations, alter physical and chemical character of streams, and change the composition of stream biota. Harvest of salmonids reduces the abundance and alters the size-and-age structure of populations. Introduced fish species can adversely affect native salmonids through competition, predation, and disruption of physical habitat. Similarly, hatchery-reared salmonids may have similar impacts as well as altering the genetic structure of populations through introgression.

1.5.1 Foresfry

Forest practices result in removal and disturbance of natural vegetation, disturbance and compaction of soils, construction of roads, and installation of culverts. Removal of vegetation typically reduces water loss to evapotranspiration, resulting in increased water yield from the watershed. In general, increases in water yield are greater west of the Cascades than they are on the east side. Increases in peak flows following logging have been reported and likely result from combined effects of vegetation removal and more rapid routing of water from uplands to the stream channel. Short-term increases in summer base flows frequently follow logging; however, evidence from one Cascade watershed suggests base flows may be reduced over the long term, particularly if coniferous vegetation is replaced by hardwood- dominated stands.

Site disturbance and road construction typically increase sediment delivered to streams through mass wasting and surface erosion, which can elevate the level of fine sediments in spawning gravels and fill substrate interstices that provide habitat for aquatic invertebrates. The removal of riparian canopy reduces shading and increases the amount of solar radiation reaching the streams, resulting in higher maximum stream temperatures and increased diel and seasonal fluctuations. In addition, the loss of riparian vegetation may increase radiative cooling during the winter, enhancing the formation of anchor ice. In other systems, increases in winter stream temperatures have been observed after logging. Increases in maximum temperature after logging depend on the size and morphology of the stream and on the type and density of canopy removed. Altered stream temperatures persist until prelogging levels of shade are re-established, which may take from less than 10 to more than 40 years.

Timber harvest removes plant biomass, and hence nutrients, but nutrients are more available to streams immediately following harvest, resulting in part from addition of slash to the forest floor, accelerated decomposition of litter, and increased runoff and erosion. This short-term increase diminishes as soils stabilize and revegetation occurs. Where logging occurs in riparian areas, delivery of leaf litter and large woody debris to the stream is reduced, and may significantly alter the nutrient balance and physical character of the stream. Loss of large woody debris, combined with alteration of hydrology and sediment transport, reduces complexity of stream micro- and macrohabitats and causes loss of pools and channel sinuosity. These alterations may persist from decades to centuries. Changes in habitat conditions may affect fish assemblage structure and diversity (e.g., favoring species that prefer riffles rather than pools), alter the age-structure of salmonid populations, and disrupt the timing of life-history events. Other effects on salmonids include reduced embryo survival and fry production, decreased growth efficiency, increased susceptibility to disease and predation, lower overwinter survival, blocked migration (e.g., inadequate culverts), and increased mortality through anglers' improved access to streams.

1.5.2 Grazing

Grazing results in the removal of natural vegetation, the alteration of plant-community composition, and the modification of soil characteristics, which in turn affect hydrologic and erosional processes. Effects are particularly acute in the riparian zone, where livestock tend to congregate, attracted by water, shade, cooler temperatures, and an abundance of high-quality forage. In general, grazed lands have less vegetation and litter cover than ungrazed lands, and in many areas of the West, perennial grasses have been replaced by non-native annual grasses and weedy species. Greater exposure of soils leads to splash erosion, which decreases soil permeability and results in more rapid runoff of precipitation to the stream channel. **As** a consequence, peak flows may be higher and summer base flows lower in watersheds that are intensively grazed.

Livestock also affect vegetation and soils through trampling. Trampling soils in arid and semi-arid lands may break up the fragile cryptogamic crust (comprised of symbiotic mosses, algae, and lichens) causing reduced infiltration, increased runoff, and reduced availability of nitrogen for plant growth. In addition, trampling detaches soil particles, accelerating surface erosion in upland areas, and may promote mass wasting along streambanks. Mass wasting also occurs where grazing has eliminated riparian vegetation and hence the root matrix that helps bind soil together. All of these processes result in increased sediment transport to streams. Animals also redistribute seeds and nutrients across the landscape, especially to riparian zones or other attractors, such as spring seeps or salt blocks. Devegetating riparian zones reduces shading and increases summer stream temperatures—often in streams that are where temperatures are near the upper limit of the tolerable range for salmonids— and may also increase the formation of anchor ice in the winter. Grazing also results in changes in channel morphology through changes in hydrology, sedimentation, and loss of bank stability. Streams in grazed areas tend to be wider and shallower, and consequently warmer in summer, than in ungrazed reaches. In some instances, streams in grazed areas incise in response to increased peak flows, effectively disconnecting the stream channel from the floodplain. Incision further alters the hydrology of the stream by lowering the water table and, consequently, the plant community occupying the riparian zone may shift from hydric (wetland) to xeric vegetation. Grazing in the riparian zone can reduce recruitment of large woody debris, especially because re-establishment of riparian shrubs and trees rarely occurs if grazing pressure is not reduced. Loss of woody debris reduces retention of gravels, creation and maintenance of pool habitats, and instream cover. General effects of grazing on salmonids include reduced reproductive success because of sedimentation of spawning gravels, alteration of food supplies through changes in primary and secondary production, reduced fish densities, and shifts in the composition of fish, invertebrate, and algal communities.

1.5.3 Agriculture

Although agriculture is not a dominant land use in the Pacific Northwest (approximately 16% of the total land area), alterations to the land surface are more severe than those caused by forestry or grazing, are generally permanent, and tend to involve repeated disturbance. Replacing natural grasslands, forests, and wetlands with annual crops leaves much area unvegetated during part of the year and dramatically changes the function of plants and soil microbes in the tilled areas. Repeated tillage, fertilization, and harvest permanently alter soil character, resulting in reduced infiltration and increased surface runoff. These changes alter seasonal streamflow patterns by increasing high flows, lowering water tables, and reducing summer base flows in streams. Channelizing to reduce local flooding and alter the geometry of cropped lands also facilitates more rapid routing of water to the stream channel, thereby increasing peak flows downstream. Sediment yield from agricultural lands is typically greater than from prairie, forest, or wetland areas, and can reduce the quality of spawning gravels and the abundance of food organisms. Where riparian shading is lost or summer base flows are reduced, stream temperatures are increased. Nutrients, insecticides, and herbicides are typically elevated in streams draining agricultural areas, reducing water quality. Channelization, snag removal, revetments, and removal of riparian vegetation reduce habitat complexity, decrease channel stability, and alter the food base of the stream. As a result, incised and channelized streams in agricultural areas typically support smaller fish and fewer fish species.

1.5.4 Urbanization

Urbanization has affected only 2% of the land area of the Pacific Northwest, but the consequences to aquatic ecosystems are severe and long-lasting. The land surface, soil, vegetation, and hydrology are all significantly altered in urban areas. **As** development proceeds, the percentage of land covered by impervious surfaces increases, reducing the area available for infiltration and increasing surface runoff. Buildings, parking lots, roads, gutters, storm drains, and drainage ditches in combination quickly shunt precipitation to receiving streams, resulting in an increased magnitude and frequency of peak discharge and reduced summer base flow. Sediment delivery typically increases during construction activities. The total vegetated area is greatly reduced, and replacement vegetation, typically lawns and ornamental plants, require water, fertilizers, and pesticides. Riparian corridors frequently are constricted, disabling or altering riparian function. Loss of riparian vegetation and reduced base flows allow greater heating of streams during summer. In addition, the lack of recruitment of large wood combined with increased erosive potential of peak flows increase scouring of the streambed and downstream transport of wood, resulting in simplified stream channels and greater instability, These effects are exacerbated when streams are channelized and where banks are reinforced with concrete, rip-rap, or other hard structures. Water quality is adversely affected by inputs of fertilizer and pesticides washed from lawns and yards, discharge from sewage treatment facilities and industrial sources, and contaminated runoff from surface streets. The highly altered streams found in most urban areas provide poor habitat for fish and other aquatic biota.

1.5.5 Mining

Sand and gravel mining in streams and on adjacent floodplains have substantial effects on stream channels and hydraulic characteristics. In addition to the immediate morphological changes in stream channels caused by excavation, channels continue to exhibit instability, accelerated erosion, and altered substrate composition and structure after extraction has ceased. Downcutting of stream channels frequently follows gravel mining, often exceeding 4–6 meters in depth over periods ranging from months to decades. The downcutting and simplification of stream channels results in increased flood peaks, increased sediment transport, increased temperatures, and decreased base flows. The most direct impacts to salmonids are degradation and simplification of spawning and rearing habitats and increased turbidity. In addition, increased turbidity and decreased substrate stability influence lower trophic levels, upon which salmonids depend for food.

Mineral mining also substantially affects aquatic ecosystems. Although hydraulic mining is uncommon today, previously degraded habitats have not yet recovered and still exhibit excessive sediment transport, downcutting, and instability. For example, hydraulic mining (e.g., gold) from stream deposits and hillslopes dramatically altered stream channels, riparian zones, and floodplains. Recovery may take generations where channels have been modified and acid drainage, radioactive materials, and metals from mining wastes contaminate streams, Increased sediments, acidification, and chronic pollution from mine wastes seriously degrade aquatic habitats throughout the West. Streams receiving chronic metal pollution typically support few or no fish and degrade invertebrate assemblages.

1.5.6 Dams and Irrigation

Hydroelectric dams, impoundments, and withdrawing water for irrigation have significantly contributed to the decline of salmonids in the Pacific Northwest. Dams have impeded or blocked passage by adult and juvenile salmonids, and have caused gross changes in habitat conditions of rivers and streams. In the Columbia River basin, an estimated 55% of the total area and **33%** of the total stream miles are no longer accessible to anadromous salmonids because of dams. At dams, injury and mortality to juveniles occurs as a result of passage through turbines, sluiceways, juvenile bypass systems, and adult fish ladders. Dams and reservoirs increase the time it takes juveniles to migrate to the ocean, which increases exposure to predation. Attempts to bypass dams by barging and trucking may facilitate transmission of parasites and disease. Below hydroelectric facilities, nitrogen supersaturation may also negatively affect migrating salmon.

Hydrologic effects of dams and withdrawals for irrigation include water-level fluctuations, altered seasonal and daily flow regimes, reduced water velocities, and reduced discharge volume. Drawdowns and diversions reduce available habitat area and concentrate organisms, potentially increasing predation and transmission of disease. Dams have eliminated many spawning areas on large river systems and have created slackwater environments that are favorable to salmonid predators, including squawfish and a host of non-native piscivores. Impoundments alter natural sediment transport processes, causing deposition of fine sediments in slackwater areas, reducing flushing of sediments through moderation of extreme flows, and decreasing recruitment of coarse material (including spawning gravels) downstream of the obstruction. Return-flows from irrigated lands tend to have high sediment content, turbidity, and pesticide and fertilizer concentrations. Impoundments and water withdrawals also change the thermal regimes of streams. Temperatures may increase in shallow reservoirs and where return-flows from irrigation have been heated. Below deeper reservoirs that thermally stratify, summer temperatures may be reduced through release of hypolimnetic waters, but fall temperatures tend to increase as heated water stored during the summer is released. These changes in water temperatures affect development and smoltification of salmonids as well as influence the success of predators and competitors and the virulence of disease organisms. Dissolved oxygen concentrations may be reduced during both summer and winter from withdrawals for irrigation. In summer, high temperatures of return-flows reduce the oxygen-holding capacity of water; in winter, drawdown of impoundments may facilitate freezing, which diminishes light penetration and photosynthesis, potentially causing fish kills through anoxia.

1.5.7 Salmonid Harvest

Although this document focuses on the effects of human activities on salmonid habitats, it is important to acknowledge the effects fisheries have had on salmonid populations in the Pacific Northwest. Commercial, recreational, and tribal harvest of salmonids by humans constitutes a significant source of mortality for both anadromous and resident species, with harvest rates of adults in many fisheries exceeding 50% to 80% or more. Adverse effects of harvest on salmonids are particularly difficult to control in mixed-stock fisheries, where multiple species, stocks, and age classes are harvested together. Strong and weak stocks are harvested at comparable rates, as are fish of wild and hatchery origin. Mixed-stock fisheries are especially detrimental to naturally small populations or populations that have been depressed by human activities.

In addition to reducing total escapement of adult salmonids, harvest alters the age- and size-structure of salmonid populations. For many populations of anadromous salmonids, particularly species that spend several years at sea, mean size and age of harvested adults have steadily declined. This occurs because immature individuals are vulnerable to troll fisheries over a number of years. Consequently, larger and older individuals are harvested at a higher rate than individuals that mature earlier and at smaller size. Changes in size structure may also result from size-selective fishing gear. Changes in average size and age of individuals influences success of salmonid populations in several ways. Large size may confer several advantages including the ability to negotiate large barriers, higher fecundity, deeper deposition of eggs (and thus reduced risk of scouring during freshets), and utilization of larger, better oxygenated spawning gravels.

Harvest of salmonids can also influence the timing of certain life history events, including adult migrations, spawning, and juvenile migrations. Selective removal of early or late migrants can result in shifts in the timing of peak migration and spawning of a population. Finally, harvest of salmonids by humans can alter the fundamental structure of stream ecosystems through reduction of nutrient inputs from salmon carcasses as populations decline and average size of fish decreases.

1.5.8 Introduced Fish and Hatcheries

Introductions of non-native fish species and artificially propagated native salmonids pose additional risks to wild salmonids. Effects of species introductions on native fishes may include elimination, reduced growth and survival, and changes in community structure. Six mechanisms allow introduced fish to dominate or displace native fish: competition, predation, inhibition of reproduction, environmental modification, transfer of new parasites or diseases, and hybridization. Introduced species may thrive best where extensive environmental modification has already occurred.

Artificial propagation of native salmonids has been used for decades to mitigate effects of habitat loss and to increase returns for harvest. Although artificial propagation may in some instances increase salmon and trout available for harvest, hatchery introductions can result in a number of unintended and undesirable consequences for wild salmon and trout populations, for example, competition between hatchery and wild fish for food, habitat, or mates. Once in the ocean, large numbers of hatchery smolts may result in density-dependent decreases in survival and growth, although evidence of density-dependent effects in ocean environments is mixed. Other adverse effects of hatchery introductions include transmission of diseases between hatchery and wild populations, alterations of fish behavior (e.g., stimulation of premature smolt migration), and increased predation on wild fish (direct predation of hatchery fish on wild fish or attraction of predators). In addition to ecological effects, introduction of hatchery fish may lead to genetic changes in wild populations, including elimination of unique genomes in local stocks, loss of genetic variability between populations, and depressed fitness where introgression occurs.

The operation of hatchery facilities may adversely affect wild salmonid populations by contributing effluent with high concentrations of nutrients or disinfectant chemicals and by introducing pathogens. Hatchery weirs or diversion structures can impede the migration of wild stocks, and diversions of water for hatchery use reduces the amount available for wild stocks. Removal of wild fish for brood stock may threaten the genetic integrity of wild stocks, particularly for small or depleted stocks. Lastly, the removal of fish for brood stock decreases the amount of nutrients available in upstream reaches because salmon carcasses are not deposited.

Hatchery supplementation has social repercussions that influence wild salmonids directly and that affect the ability of managers to restore salmonid populations. Hatchery supplementation increases harvest pressure on wild populations in mixed-stock and terminal fisheries, particularly during years when survival of hatchery fish is low due to poor environmental conditions, and fisheries become overcapitalized. In addition, once commercial and sport fishers have invested large sums of money in fishing gear, they may resist increased fishing restrictions, making it difficult for managers to enact stricter protection for wild stocks. Finally, the long history of hatchery programs in the United States has instilled a perception in the public that habitat losses or degradation can be mitigated through artificial propagation, a perception that may impede implementation of more ecologically sound restorative activities.

1.6 Effects of Atmospheric and Ocean Circulation

Marine productivity depends on atmospheric and oceanic circulation and strongly affects abundance of salmonids and other fishes. Surface currents of the northeastern Pacific are dominated by the "West Wind Drift," which flows west-to-east across the Pacific and bifurcates as it approaches North America into the Alaska Current flowing north and the California Current flowing south. Changes in climatic conditions affect the behavior of the West Wind Drift. In years where a strong Aleutian Low Pressure system develops off the south coast of Alaska, typical of El Niiio conditions, a greater percentage of cold, nutrient-rich water is diverted north into the Alaska Current. When the Aleutian Low is weaker, typical of La Niiia years, more water from the West Wind Drift is diverted south towards California. These shifts, combined with changes in prevailing wind directions and upwelling patterns, can substantially affect conditions for salmonids entering the ocean. Changes in surface currents and upwelling strength influence temperature, salinity, and nutrients, thereby affecting the abundance of food available to juvenile salmonids, the number and distribution of predators and competitors, and the transport of smolts entering the ocean (along-shore versus off-shore). Recent evidence suggests that when ocean conditions are poor for salmonids in the Pacific Northwest, conditions are favorable to Alaskan stocks and vice versa.

Cycles in marine productivity can mask the effects of habitat degradation in freshwater environments or other stressors of salmonid populations. Long-term trends in the ability of freshwater environments to support salmonids may not be evident during periods of favorable oceanic conditions, particularly for populations augmented by hatchery fish. However, as ocean conditions shift towards less favorable conditions (particularly for hatchery fish), increasing pressure from overcapitalized fisheries can dramatically reduce the abundance of wild stocks.

1.7 Practices For Restoring and Protecting Salmonids and Their Habitats

Virtually all land-use and water-use practices have some effect on aquatic ecosystems, as do the harvest of salmonids and the introduction of non-native and hatchery fish. However, there are numerous opportunities, through planning and specific practices, for minimizing these effects or mitigating for past damage. Regardless of the activity, emphasis should be placed on preventing (rather than mitigating) damage, particularly in those areas where high-quality habitats and stable salmonid populations remain.

Impacts of harvest on wild salmonids can be best controlled through terminal and bay fisheries that target adults as they return to their natal streams.

Terminal fisheries provide greater protection for weak stocks by targeting hatchery runs instead of wild stocks, by allowing late-maturing fish to reach maturity, and by reducing the incidental mortality of subadults. These targeted fisheries avoid problems associated with managing for indicator or weak stocks traditionally used in open ocean, mixed-stock fisheries. Harvest methods can also be changed to target hatchery stocks and reduce incidental mortality of wild populations. Traps, fish wheels, and hook-and-line angling all cause lower mortality than gillnets or trolling. Special sport angling restrictions, including catch-and-release angling, minimum size or slot limits, and bag limits may further reduce mortality or minimize size-selective harvest. Accurate monitoring of escapement levels of specific stocks is essential for establishing exploitation levels that ensure the long-term persistence of individual stocks.

Growing evidence of the adverse ecological, genetic, and social consequences of hatchery operations suggests substantial modification, curtailment, or elimination of hatchery programs for salmonids would benefit wild populations, though not without adverse short-term social and economic impacts. Emphasis of hatchery programs is beginning to shift from increasing fish harvest to conserving endangered species or supplementing weak stocks, though the risks of using hatcheries for these purposes are still being debated. Potentially beneficial hatchery programs include those to re-establish native species into waters where fish have been extirpated by human activities; those to sustain a presently overharvested fishery through a planned program of downsizing and transition to other employment or from reliance on hatchery fish to reliance on wild fish; and those to augment weak stocks (put-and-grow stocking) in waters having little or no reproductive habitat but substantial productive potential where stocking will not harm indigenous biota, Impacts of introduced (non-native) fish species on wild salmonids can be minimized by ceasing the stocking of non-native fish into waters that contain wild salmonids, by direct removal by piscicides and electrofishing, and by indirect removal through use of unrestricted catch limits. Restoring streams and rivers to their natural temperature and flow regimes may reduce the spread of non-native species into salmonid streams.

A number of large-scale habitat restoration programs are currently underway or in the planning stages. In the Kissimmee River, Florida, steps are being taken to re-establish natural channel configuration, floodplains, and hydrologic regimes. In the Elwha (Washington) and Rogue River (Oregon) basins, dam removal has been proposed to restore salmon habitats and remove barriers to migration. Elsewhere, the impacts of dams are being reduced by assuring instream flows, especially at critical times; screening turbine intakes; and improving bypass systems. Direct impacts to river channels can also be minimized by retaining large woody debris and channel complexity and by restricting snagging and channelization.

Impacts of forest practices can be reduced through longer rotations; selective harvesting instead of clear-cutting; logging during the dry season or when the ground is frozen; use of high lead, skyline, and helicopter logging instead of ground-based equipment; use of designated skid trails; minimizing site-preparation practices that compact or scarify soils; retention of riparian buffer zones along streams; designation of no-cut zones in areas prone to mass failures; careful placement and maintenance of roads; and decommissioning and reseeding of roads when logging is completed. These activities function to minimize the percentage of the watershed in a disturbed state, reduce the total area of ground disturbance and soil compaction, minimize surface runoff and sediment loads, and protect and preserve the function of riparian zones.

The effects of range practices can be reduced by resting pastures, decreasing numbers of livestock, controlling livestock distribution through fencing of riparian zones or watering of stock away from riparian areas, controlling forage use, controlling season of use, and determining the kind of livestock best suited for the area. These practices can serve to reduce grazing stress, ensure that sufficient vegetative cover remains after the grazing season, promote the re-establishment of riparian vegetation (particularly woody shrubs and trees), and keep stock out of riparian zones, although site-specific conditions will determine their relative effectiveness.

Agricultural practices and policies that promote water and soil conservation and that reduce chemical application can all reduce effects on aquatic ecosystems. Examples include switching to crops that do not require irrigation, ditch lining and drip irrigation, screening of intakes for irrigation, increasing vegetative cover (e.g., permanent rather than annual crops), conservation tillage, planting grass in water ways (for soil conservation), organic farming, integrated pest management, and increasing tax relief for farmers employing conservation practices as well as penalties for those who do not.

Most of the impacts of gravel mining relate to changes in channel morphology that create channel instability, cause bedload movement, and increase sedimentation. Consequently, these effects can be most productively reduced by eliminating instream mining. Bar scalping instead of below-surface extraction has been used to minimize turbidity and direct damage to spawning habitats; however, changes in channel morphology are likely to occur as water levels rise. Abandoned gravel mines in floodplain areas may offer opportunities for increasing off-channel habitats. Effects of mineral mining can be reduced by burying toxic materials below the root zone, by rehabilitating the site using created natural contouring and re-established natural vegetation, and by controlling mining-generated solids and liquids with containment structures and waste treatment.

Urbanization permanently alters many natural watershed processes, and in some cases, little may be done to mitigate effects. Thus, the most effective way to minimize impacts is through careful land-use planning that minimizes the total impervious area and that precludes development along streams and in natural floodplains. Sewage treatment and programs to foster water conservation, minimize chemical applications, and prevent toxic materials from being dumped into drainage structures can reduce impacts of urbanization to water quality.

Finally, because the condition of aquatic habitats is ultimately tied to resource consumption—the use of water, electricity, wood products, meat and wool, food and nonfood crops, and mineral resources—conservation of salmonids will require re-examination of fundamental aspects of our culture, including actions of individuals, population and economic policies, and ethical concerns. Policies that promote conservation need to be encouraged while those that foster waste and overconsumption need to be discouraged. Education is central to increasing the awareness of citizens as to how their actions directly or indirectly affect salmonids and their habitats.

1.8 Relevant Federal Laws for Protecting and Restoring Salmonid Ecosystems

Several federal laws, notably the Clean Water Act (CWA), the Endangered Species Act (ESA), the National Environmental Policy Act (NEPA), and the Food Security Act (FSA), are or could be employed to protect aquatic and riparian habitats on nonfederal lands. The goals of the CWA are to restore and maintain the chemical, physical, and biological integrity of the Nation’s waters; to eliminate discharge of pollutants into waters; to attain water quality that provides for the protection and propagation of fish, shellfish, and wildlife; and to develop and implement area-wide waste treatment management to control pollutant sources. The NEPA declares a national policy that encourages harmony between humans and their environment, reduces environmental damage, and improves understanding of ecological systems. The **ESA** seeks to conserve the ecosystems upon which threatened and endangered species depend and to provide a program to conserve listed species and their ecosystems. The FSA encourages conservation by making ineligible for Federal price supports, loans, crop insurance, or disaster payments any landowner who produces a crop on highly erodible lands or on converted wetlands. Each of these laws may be used to provide Federal leadership in furthering the goals of habitat conservation.

1.9 Monitoring Conservation Efforts

The success of salmonid conservation efforts depends on a rigorous monitoring program for determining whether conservation plans are being implemented and if they are effective. Examples drawn from existing programs to monitor wetland permits, forest plans, point-source discharges, and rural best-management plans indicate a number of common shortcomings. These include inadequate funds and staff, unclear objectives and criteria, failure to use remote sensing and site visits, and lack of computerized data systems. Periodic status reports and peer reviews are essential to successful monitoring as well. To make a monitoring program most useful and cost-effective, it must be regional, have a statistical design, and be based on quantitative physical, chemical, and biological indicators. Consistency of indicators between Federal and State monitoring programs is also essential.

2 Introduction

The structure of aquatic ecosystems—the physical habitats, the material and energy resources, and the associated biological communities—arises from complex interactions among numerous processes that occur in upland areas, within riparian zones, and in stream channels, lakes, or estuaries. Physical processes act in concert with vegetative characteristics to provide the physical and chemical context within which aquatic systems develop and to regulate the exchange of material and energy from the watershed to the stream channel. Biological processes both influence the conversion of material and energy as well as govern the relationship of organisms to one another and to their environment. Collectively, physical, chemical, and biological processes give rise to ecosystem structures, which in turn exert influence back on those processes. These ecosystem-shaping processes operate over a wide range of temporal and spatial scales. Protection and recovery of salmonid habitats fundamentally depends on maintaining and restoring, in both space and time, the natural rate or frequency of occurrence of these processes and the ecosystem structures to which they give rise.

Part I of this document comprises a technical foundation for understanding salmonid conservation principles and developing salmonid conservation plans in an ecosystem context. We intentionally focus on freshwater habitats but recognize that many other natural and anthropogenic factors, which we only discuss briefly, influence greatly the abundance of salmonids, including fish harvest, hatchery practices, habitat conditions in near-shore areas, and natural variation in ocean productivity. Conclusions are based on our assessment of the scientific literature. Because some topics are thoroughly discussed in this literature and others are not, certain sections of the document are relatively complete and robust, but others are more sparse.

2.1 Scope

Geographically, the scope of this document is limited to the Pacific Northwest region, including portions of California, Oregon, Washington, and Idaho that presently support or historically supported salmonid populations. Many general concepts and processes examined, however, are equally relevant outside this region. Discussion of specific habitat requirements is restricted to salmonid species that are endemic to the Pacific Northwest (Table 2-l), including the five Pacific salmon (chinook, coho, sockeye, chum, and pink salmon), trout and char with both resident and anadromous forms (rainbow and cutthroat trout, Dolly Varden char), and strictly resident species (bull trout, mountain whitefish).

In the remainder of Chapter 2, we discuss evidence of widespread declines in salmonid abundance that indicate region-wide degradation in habitat quality and ecosystem condition. We then identify strategies for restoring salmonid habitats.

Table 2-1. Common and scientific names of salmonids native to the Pacific Northwest.

Common name	Scientific name
Pink salmon	*Oncorhynchus gorbuscha* (Walbaum)
Chum salmon	*Oncorhynchus kefa* (Walbaum)
Coho salmon	*Oncorhynchus kisufch* (Walbaum)
Sockeye salmon	*Oncorhynchus nerka* (Walbaum)
Chinook salmon	*Oncorhynchus fshawyfscha* (Walbaum)
Mountain whitefish	*Prosopium williamsoni* (Girard)
Cutthroat trout	*Oncorhynchus clarki* (Richardson)
Rainbow and steelhead trout	*Oncorhynchus mykiss* (Walbaum)
Bull trout	*Salvelinus confluenfus* (Suckley)
Dolly Varden	*Salvelinus malma* (Walbaum)

These strategies emphasize the importance of maintaining natural watershed processes, providing for the diverse life-history requirements of salmonids, and re-establishing connectivity between salmonid habitats across the landscape.

In Chapters 3 and 4 we review physical, chemical, and biological processes that occur within watersheds, that influence the quality and quantity of available salmonid habitat, and that need to be maintained to ensure the persistence of salmonid stocks. Some physical and chemical processes (Chapter 3) shape stream habitats over long time periods (e.g., glaciation, volcanism) and others operate in relatively short time scales (e.g., floods, droughts, landslides). Biological processes (Chapter 4) encompass those occurring at the level of the individual organisms (e.g., physiology, behavior), populations (e.g., life history, adaptation), and communities (e.g., disease, predation, parasitism, competition). Chapters 3 and 4 provide sufficient detail about ecological processes that the effects of anthropogenic disturbances on salmonids and their habitat can be understood and evaluated. Chapter 5 describes habitat requirements specific to each stage of the salmonid life history and general characteristics of healthy aquatic and riparian systems, including physical habitat structure, streamflow, stream temperature, water quality, and important biological elements.

In Chapter 6, we discuss the effects of human activities on watershed processes and the resulting impacts on salmonids and their habitats. The discussion focuses on effects of land-use practices including forestry, livestock grazing, agriculture, mining, and urbanization. Effects of water uses, including hydroelectric dams and irrigation impoundments, are also reviewed. Although the effects of other human activities such as fish harvest, hatchery supplementation, and introduction of non-native species are largely outside the scope of this project, these issues are discussed to provide an appropriate context from which to view habitat-related issues. Chapter 7 briefly reviews general circulation patterns and the dominant physical processes controlling conditions in the northeastern Pacific Ocean. This chapter also discusses how ocean conditions influence abundance and distribution of aquatic organisms, including anadromous salmonids, and the relevance of these natural production cycles to the conservation of freshwater habitats of salmonids.

Chapter 8 identifies management systems and practices that are designed to minimize effects of human activities on salmonid habitats, with emphasis on forestry, range, and agricultural practices, as well as urban planning. Chapter 9 summarizes four Federal laws and associated amendments that pertain to conserving and protecting aquatic species and their habitats on nonfederal lands. These include the Endangered Species Act (ESA), the Clean Water Act (CWA), the National Environmental Policy Act (NEPA) and the Food Security Act (FSA).

The importance of a rigorous implementation and monitoring program for aquatic resource conservation is discussed in Chapter 10, wherein limits and inadequacies of previous programs are used as examples. The value of monitoring several physical, chemical, and biological indicators is also discussed.

2.2 Historical Background and Evidence of Habitat Degradation

Many Pacific salmon stocks have been depleted to the point that continued declines will likely result in additional listings under the Endangered Species Act (ESA) or local extirpations. Although ample evidence documents historical declines in Pacific salmonids (Ebel et al. 1989), the landmark paper by Nehlsen et al. (1991) alerted both scientists and the public to the extent of these declines. Summarizing the status of Pacific salmon of Washington, Oregon, Idaho, and California, Nehlsen et al. (1991) listed 106 stocks (unique populations) that have been extirpated from the region and 214 stocks that are at high or moderate risk of extinction or of special concern. Huntington et al. (1996) concluded that only 99 stocks of native anadromous salmonids in the region have populations greater than one-third their historical abundance, and just 20 stocks are at levels greater than two-thirds of their former abundance.

Since 1985, tribes, professional fishery societies, and conservation organizations have petitioned the National Marine Fisheries Service (NMFS) to list 24 stocks as threatened or endangered. To date four of these have been listed (Sacramento River winter chinook, Snake River sockeye, and Snake River spring/summer and fall chinook salmon). In addition, NMFS has recommended listing coho salmon as threatened throughout all of California and most of Oregon (NMFS 1995a). NMFS has added coho stocks in southwestern Washington and Puget Sound to the candidate species list; information is currently insufficient to warrant listing, but specific risk factors have been identified, and concerns need to be resolved before a final status determination is made. Commercial and sport ocean-harvests in the Pacific Northwest have been sharply curtailed in recent years because of dwindling numbers of salmon and concern for wild salmon stocks. In 1994, the ocean commercial **and** recreational fisheries for coho and chinook salmon were completely closed from Washington to Cape Falcon, Oregon, with the exception of a limited treaty Native American troll fishery for chinook salmon off northern Washington.

South of Cape Falcon to central California, fishing for coho salmon was restricted to recreational catch, and no retention of coho salmon was allowed after May 1 (PFMC 1995). In 1995, the commercial and recreational coho salmon fishery was completely closed, as was the chinook fishery from northern Washington to Cape Falcon. The fishing seasons for chinook salmon were severely restricted from Humbug Mountain to central California.

A number of factors have been implicated in the decline of Pacific salmonids including dams, overexploitation, disease, natural predation, artificial propagation, climatic variation, and the destruction and alteration of habitat. The relative importance of each of these factors in influencing salmonid populations varies across the region; however, habitat loss and modification are believed to be the major factors determining the current status of salmonid populations (FEMAT 1993). Nehlsen et al. (1991) concluded that present or future habitat degradation (including mainstem passage and flow problems) represents a significant threat to 90% of populations of anadromous Pacific salmonids identified as at high or moderate risk of extinction or of special concern. Similarly, Miller et al. (1989b) reported that physical habitat degradation was identified as a causal factor in 73% of fish species extinctions in North America during the past 100 years. A wide range of land- and water-use practices have contributed to the degradation of aquatic habitats, including timber harvesting, livestock grazing, agriculture, mining, urbanization, road construction, and construction of dams for hydroelectric power, irrigation, and flood control. Alterations in riverine systems that result from these activities include 1) changes in water quantity or flow because of water storage and irrigation or other withdrawals; 2) direct modification of channel morphology and riparian ecosystems by dams, reservoirs, channelization, draining and filling of wetlands, and dredging for navigation; 3) land-use practices that alter upland and riparian vegetation and, thus, the delivery or water, sediment, organic matter, and nutrients to streams; and 4) excessive point and nonpoint source pollution (Doppelt et al. 1993). Over time, land-use practices have substantially decreased the physical and biological complexity of ecosystems, thereby diminishing the ability of ecosystems to self repair when perturbed (FEMAT 1993).

Regional patterns in declines of salmonids and other fishes in the Pacific Northwest suggest that deterioration of freshwater habitats is widespread, with certain regions being particularly degraded. The 214 at-risk salmon stocks identified by Nehlsen et al. (1991) are distributed throughout Washington, Oregon, California, and Idaho. At least two-to-three species of fish (including nonsalmonids) are extinct or at risk of extinction in most areas of the Pacific Northwest, indicating that species losses are not isolated occurrences (Figure 2-1) (Frissell 1993b). Nawa (1994) examined population trends for 228 stocks of spring and fall chinook salmon over the period from 1940–1993 and found that 34% were extinct or nearly extinct, 24% were declining, and only 8% were not declining. Other chinook stocks were either hatchery-influenced or had unknown status. Bisson et al. (1992b) report that more Alaskan stocks of chinook, coho, chum, and steelhead increased than decreased from 1968 to 1984 (Figure 2-2). Conversely, in Washington, the Columbia River Basin, and coastal Oregon and California, declining stocks outnumbered increasing stocks for all four species examined, though the majority of stocks exhibited no significant trend over the 16-year period (Figure 2-2). Frissell (1993b) examined native fish taxa that are considered extinct, endangered, or threatened in drainage basins of the Pacific Northwest and reported a north-south gradient in the degree of endangerment (Figure 2-1); mean percentages of the native taxa considered to be extinct or at risk of extinction were 13.5% in Washington, 33.0% in Oregon, and 48.0% in California. This pattern is largely influenced by the basin-specific populations of seven widely distributed species of anadromous salmonids rather than locally endemic species such as suckers, pupfishes, and minnows. Another status review of the five Pacific salmon and the anadromous steelhead and cutthroat trout (TWS 1993) indicates a similar latitudinal gradient in the degree of endangerment for most of these species (Figure 2-3 through Figure 2-11). The general north-south gradient in salmonid declines likely reflects several factors. First, the environments in the southern portion of the salmonids' range are more extreme, with specific habitat attributes (e.g., temperature, streamflow) approaching the tolerable limits for the species. Second, there has generally been a higher degree of habitat modification in the southern part of the range. And finally, the influence of changing oceanic conditions varies with latitude (see Chapter 7).

In addition to the north-south gradient in species declines, several subregions and localized areas have an especially high degree of species endangerment. The risk of extinction is greatest in the upper Columbia—with multiple large hydropower dams and large-scale water diversions—as well as in many other undammed coastal and Puget Sound streams (Figure 2-12). Prior to development, 10–16 million salmon returned to the Columbia River to spawn each year; however, recent estimates suggest that fewer than 0.5 million wild fish now spawn in the Columbia River and its tributaries (NPPC 1992b). Coho salmon historically were abundant throughout

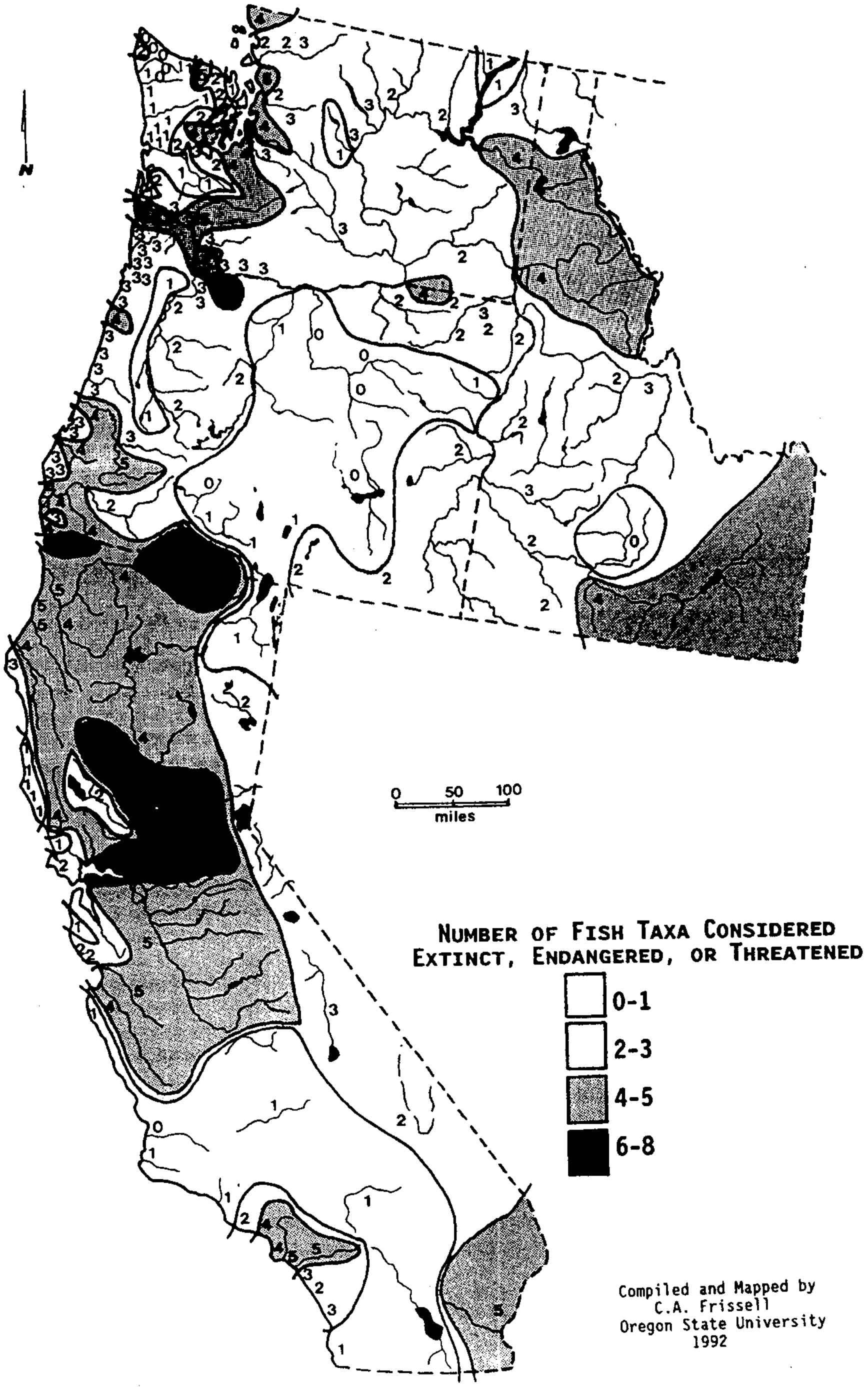
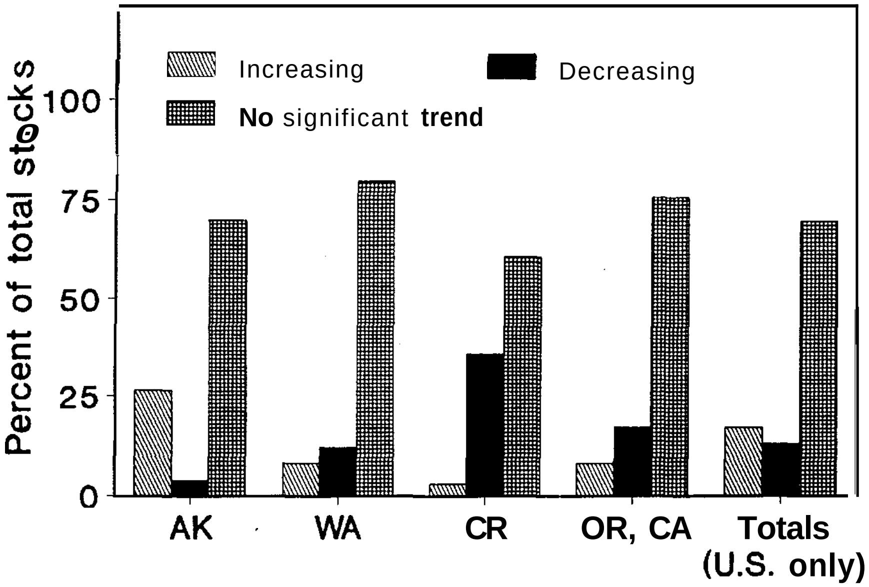
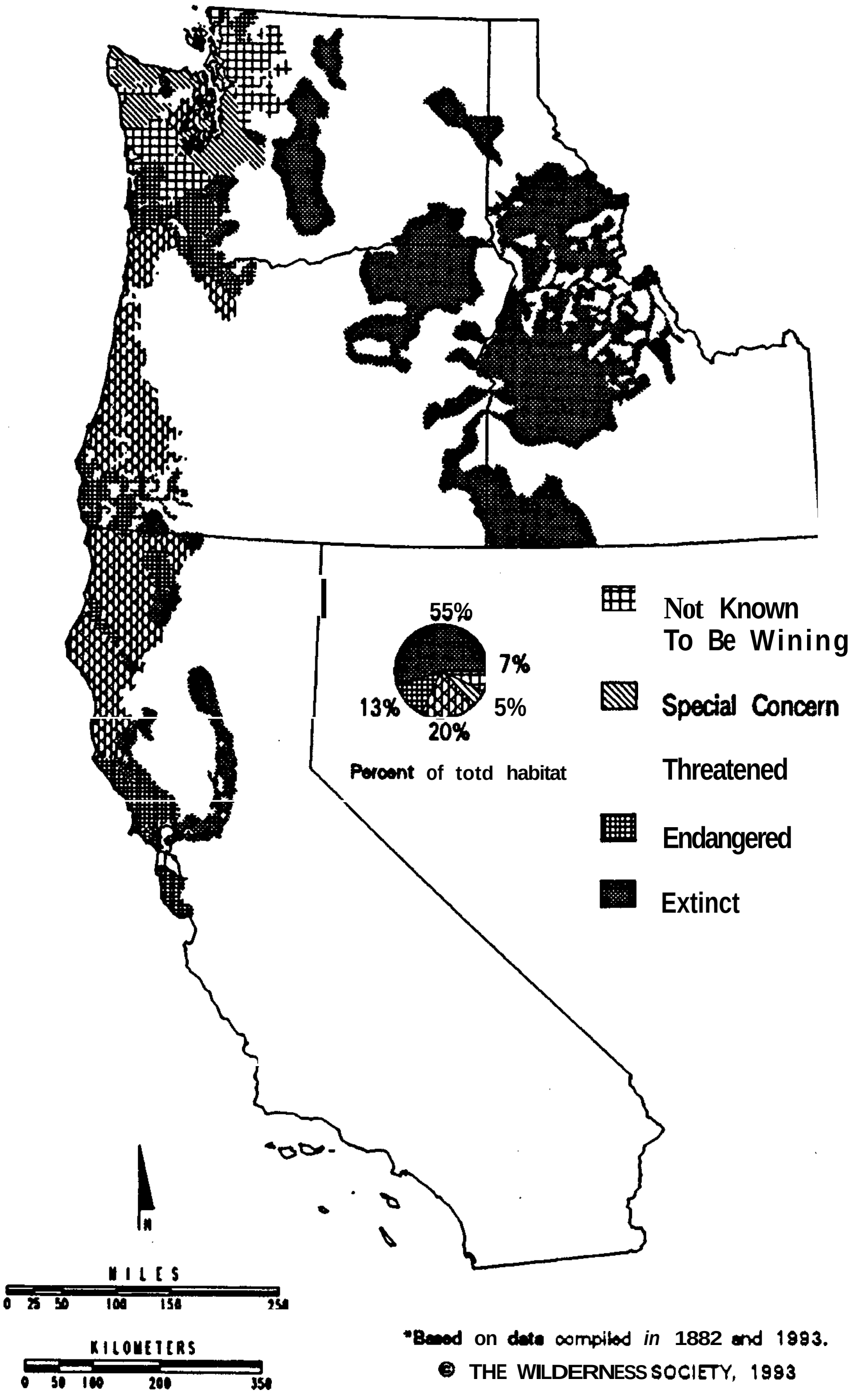

N
0 50 100
miles
NUMBER OF FISH TAXA CONSIDERED EXTINCT, ENDANGERED, OR THREATENED
0-1
2-3
4-5
6-8
Compiled and Mapped by
C.A. Frissell
Oregon State University
1992

Figure 2-1. Number and location of fish species considered extinct, endangered, or threatened in the Pacific Northwest and California. From Frissell (1993b). Reprinted by permission of Blackwell Scientific Publications, Inc.



Region	Chinook UP	Chinook DN	Coho UP	Coho DN	Chum UP	Chum DN	Steelhead UP	Steelhead DN
Alaska	43%	1%	15%	11%	3%	13%	17%	0%
Coastal WA	12%	32%	9%	0%	6%	15%		
Columbia R. Basin	3%	39%	0%	45%	0%	33%	8%	25%
Coastal OR, CA	19%	12%	2%	17%	11%	11%	20%	40%
U.S. Total	20%	22%	6%	17%	4%	14%	11%	23%

Figure 2-2. Trends in the abundance of wild stocks of chinook salmon (Oncorfiynchus tshawytscha), coho salmon (*O.* kisutch), chum salmon (*O.* kefa), and steelhead (O. mykiss) from river systems along the Pacific coast. UP = percentage of stocks significantly increasing, DN = percentage of stocks significantly decreasing. From Bisson et al. (1992b) based on data from Konkel and Mclntyre (1987). Reproduced with permission of the publisher.


55%
7%
5%
20%
13%
Percent of totd habitat
Not Known To Be Wining
Special Concern
Threatened
Endangered
Extinct
N
MILES
0 25 50 100 150 250
KILOMETERS
0 50 100 200 350

*Based on data compiled in 1882 and 1993.

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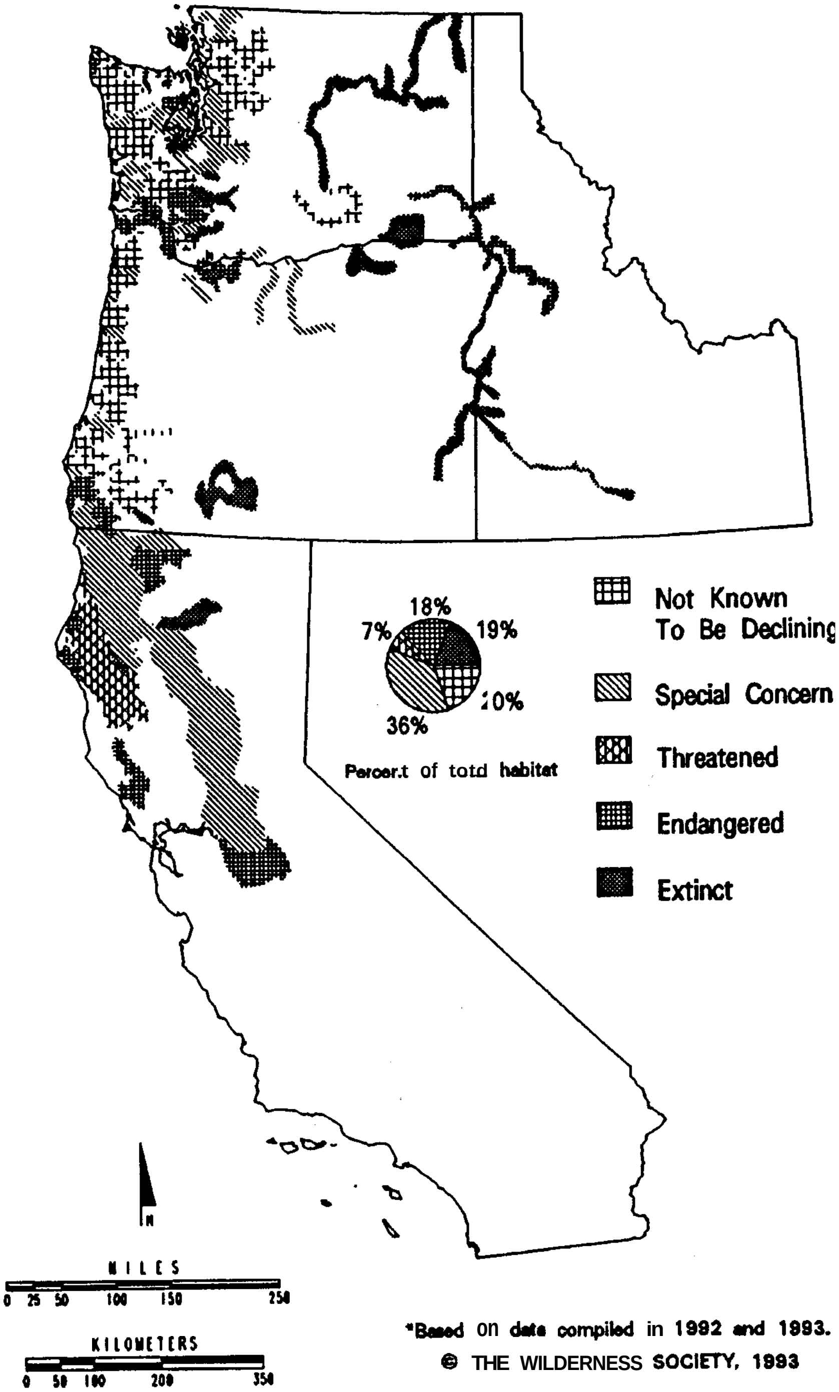
Figure 2-3. Status of coho salmon in the Pacific Northwest and California. From TWS (1993). Reproduced with permission of the publisher.


18%
7%
19%
20%
36%
Percent of total habitat
Not Known To Be Declining
Special Concern
Threatened
Endangered
Extinct
N
MILES
0 25 50 100 150 250
KILOMETERS
0 50 100 200 350

*Based on data compiled in 1992 and 1993.

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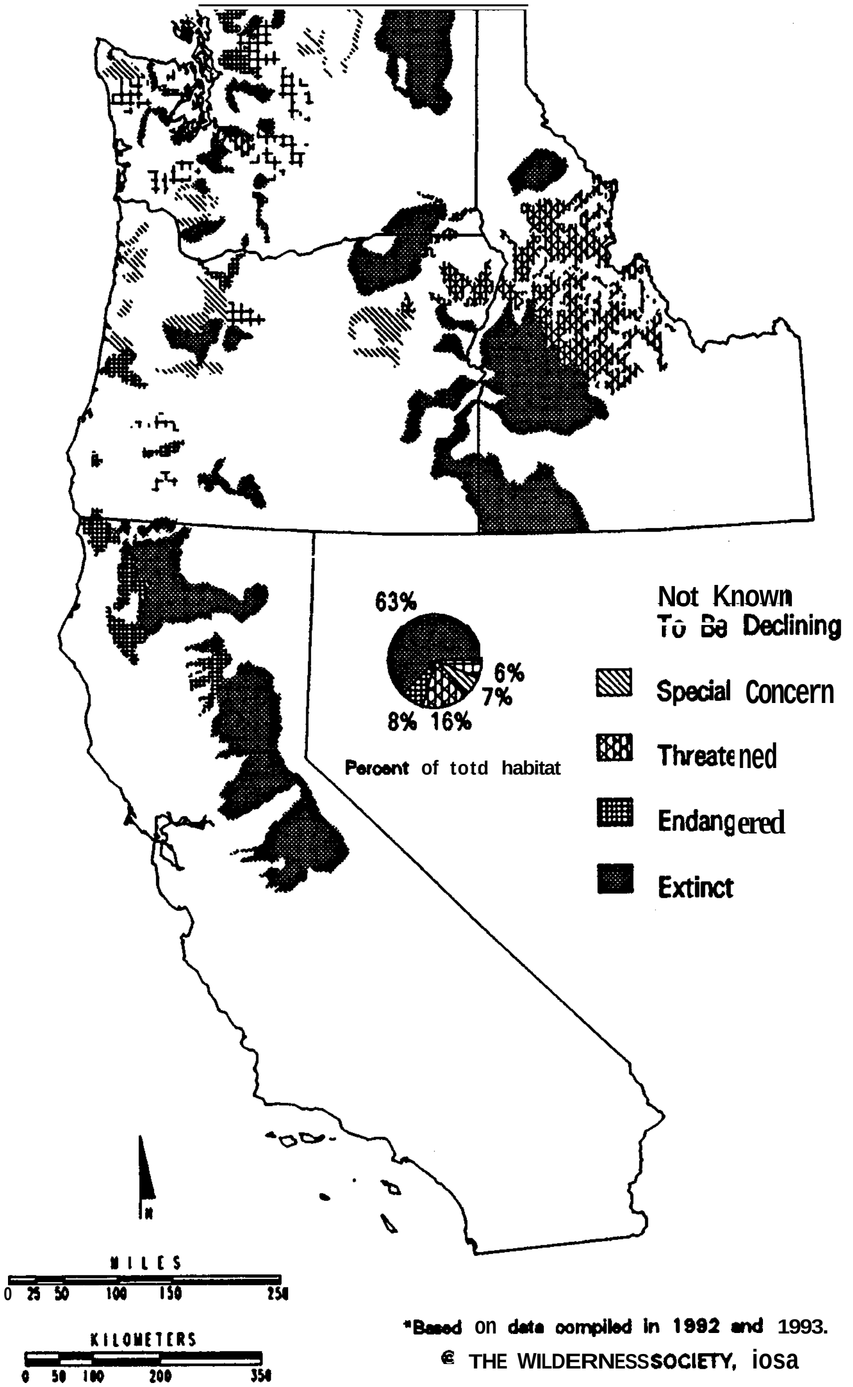
Figure 2-4. Status of fall chinook salmon in the Pacific Northwest and California. From TWS (1993). Reproduced with permission of the publisher.


63%
6%
7%
8% 16%
Percent of total habitat
Not Known To Be Declining
▧ Special Concern
▦ Threatened
▦ Endangered
■ Extinct
N
MILES
0 25 50 100 150 250
KILOMETERS
0 50 100 200 350

*Based on data compiled in 1992 and 1993.

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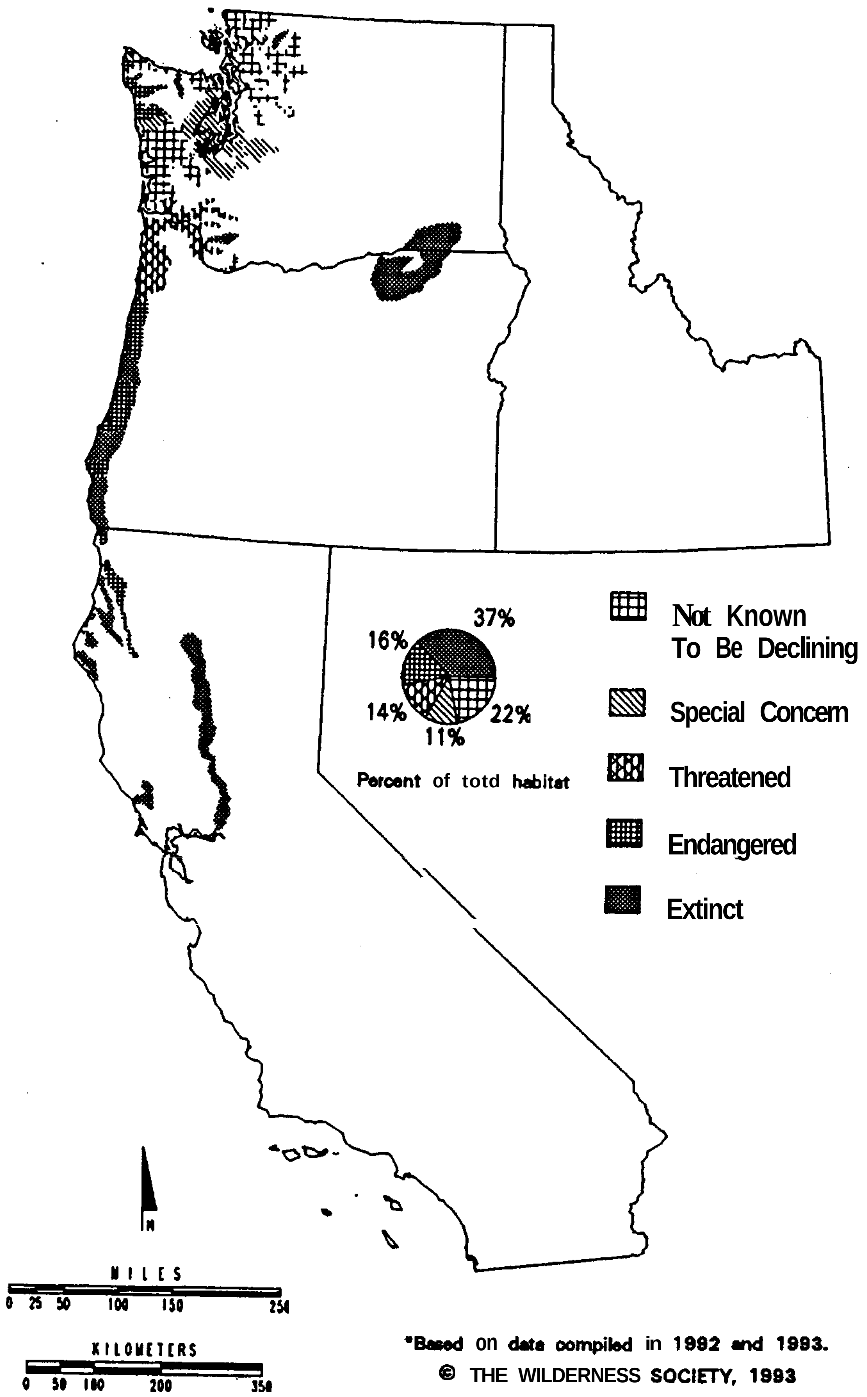
Figure 2-5. Status of spring and summer chinook salmon in the Pacific Northwest and California. From TWS (1993). Reproduced with permission of the publisher.


37%
16%
14%
22%
11%
Percent of total habitat
Not Known To Be Declining
Special Concern
Threatened
Endangered
Extinct
N
MILES
0 25 50 100 150 250
KILOMETERS
0 50 100 200 350

*Based on data compiled in 1992 and 1993.

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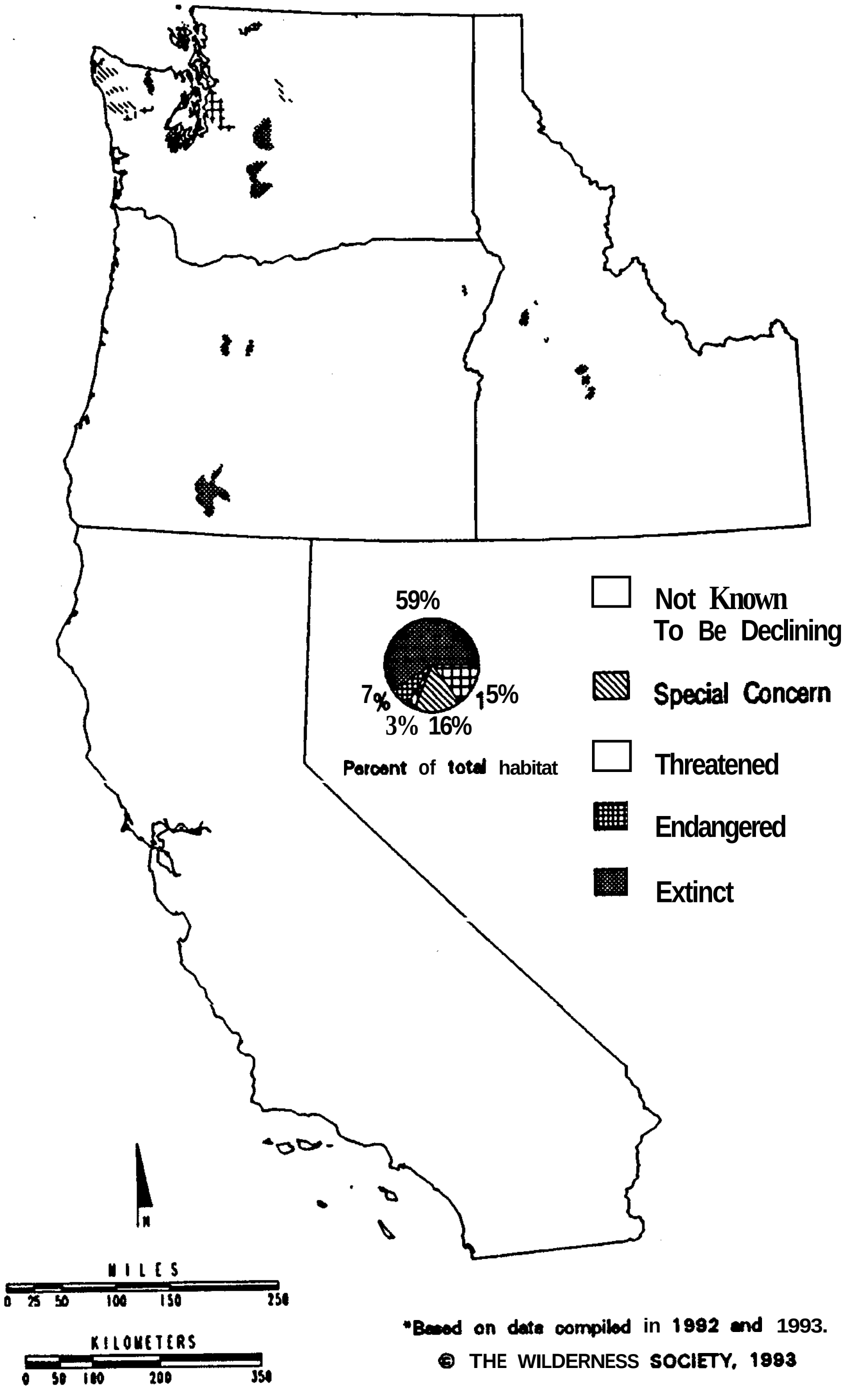
Figure 2-6. Status of chum salmon in the Pacific Northwest and California. From TWS (1993). Reproduced with permission of the publisher.


59%
7%
15%
3% 16%
Percent of total habitat
☐ Not Known To Be Declining
▨ Special Concern
☐ Threatened
▦ Endangered
■ Extinct
N
MILES
0 25 50 100 150 250
KILOMETERS
0 50 100 200 350

*Based on data compiled in 1992 and 1993.

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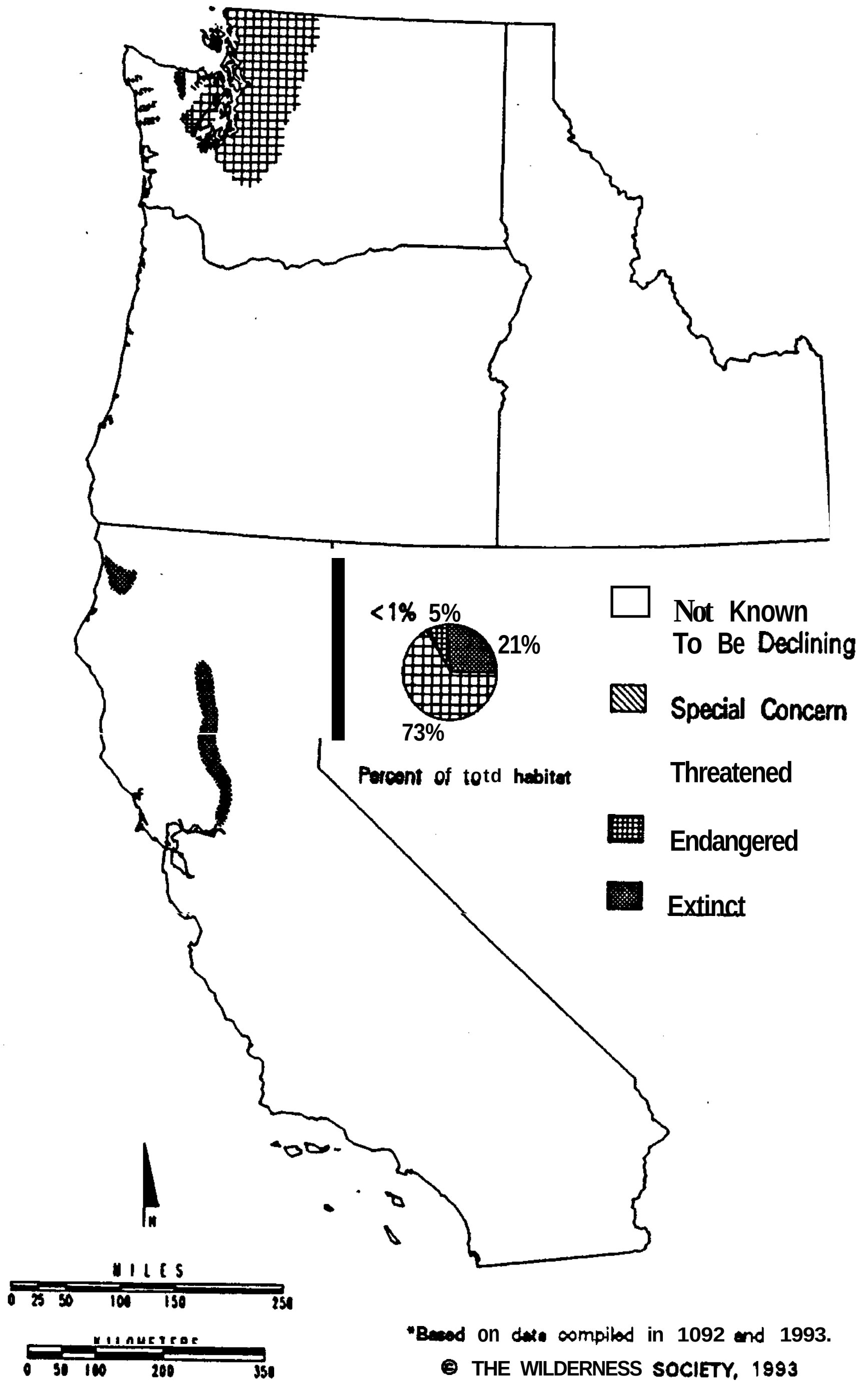
Figure 2-7. Status of sockeye salmon in the Pacific Northwest and California. From TWS (1993). Reproduced with permission of the publisher.


<1% 5%
21%
73%
Percent of total habitat
☐ Not Known To Be Declining
▧ Special Concern
Threatened
▦ Endangered
■ Extinct
N
MILES
0 25 50 100 150 250
KILOMETERS
0 50 100 200 350

*Based on data compiled in 1092 and 1993.

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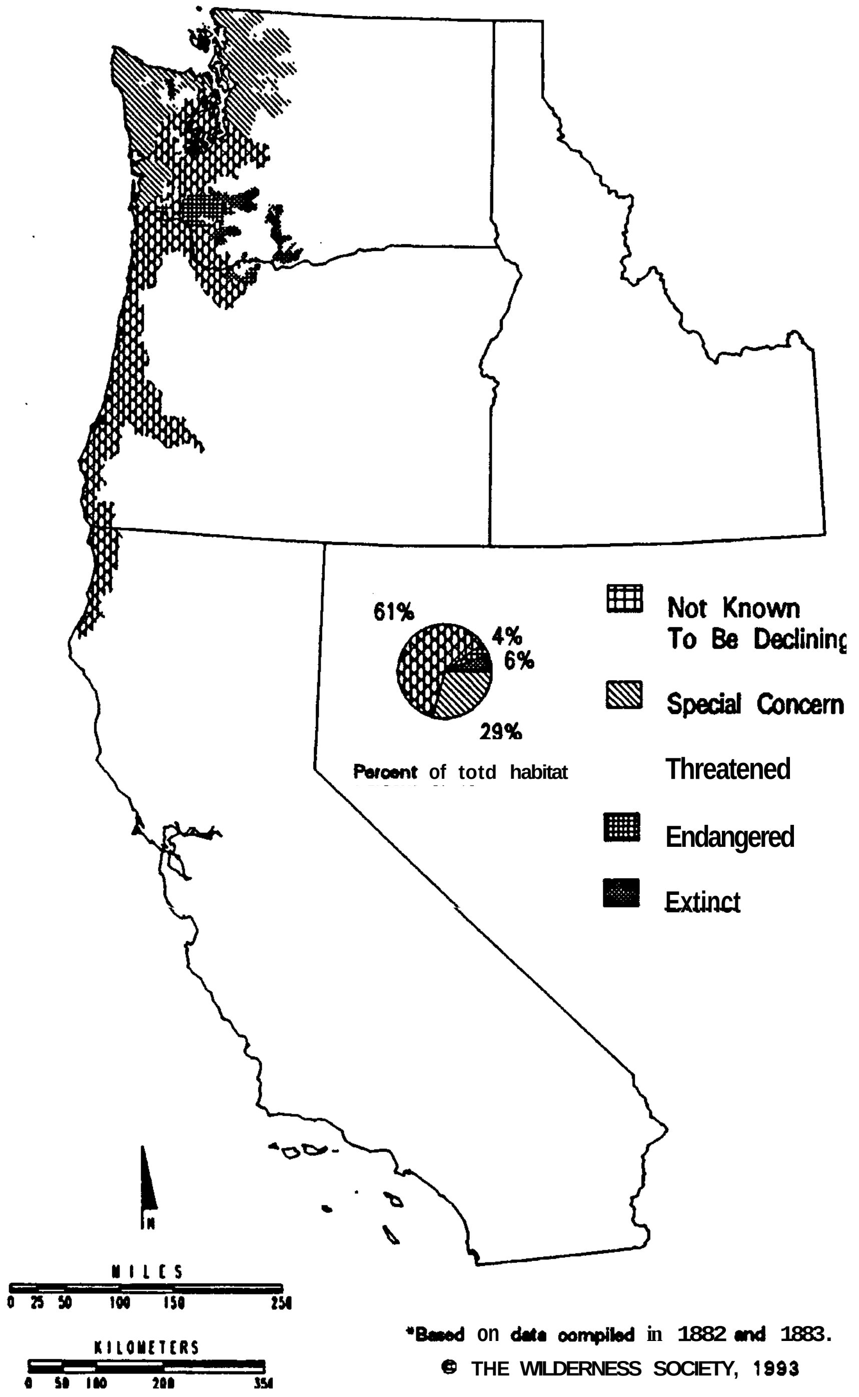
Figure **2-8.** Status of pink salmon in the Pacific Northwest and California. From TWS (1993). Reproduced with permission of the publisher.


61%
4%
6%
29%
Percent of total habitat
▦ Not Known To Be Declining
▨ Special Concern
Threatened
▦ Endangered
■ Extinct
N
MILES
0 25 50 100 150 250
KILOMETERS
0 50 100 200 350

*Based on data compiled in 1882 and 1883.

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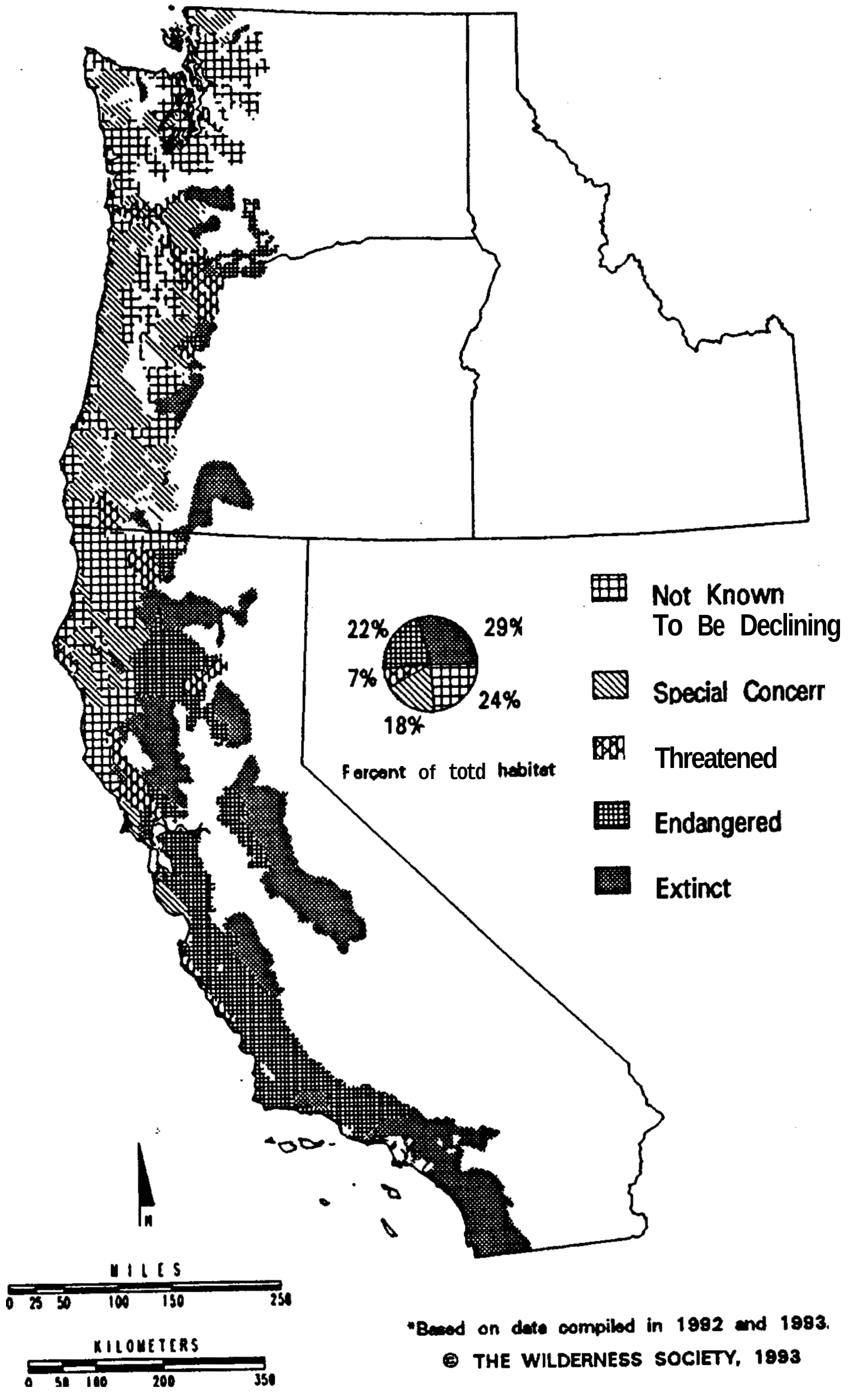
Figure 2-9. Status of sea-run cutthroat trout in the Pacific Northwest and California. From TWS (1993). Reproduced with permission of the publisher.


22%
29%
7%
24%
18%
Percent of total habitat
Not Known To Be Declining
Special Concern
Threatened
Endangered
Extinct
N
MILES
0 25 50 100 150 250
KILOMETERS
0 50 100 200 350

*Based on data compiled in 1992 and 1993.

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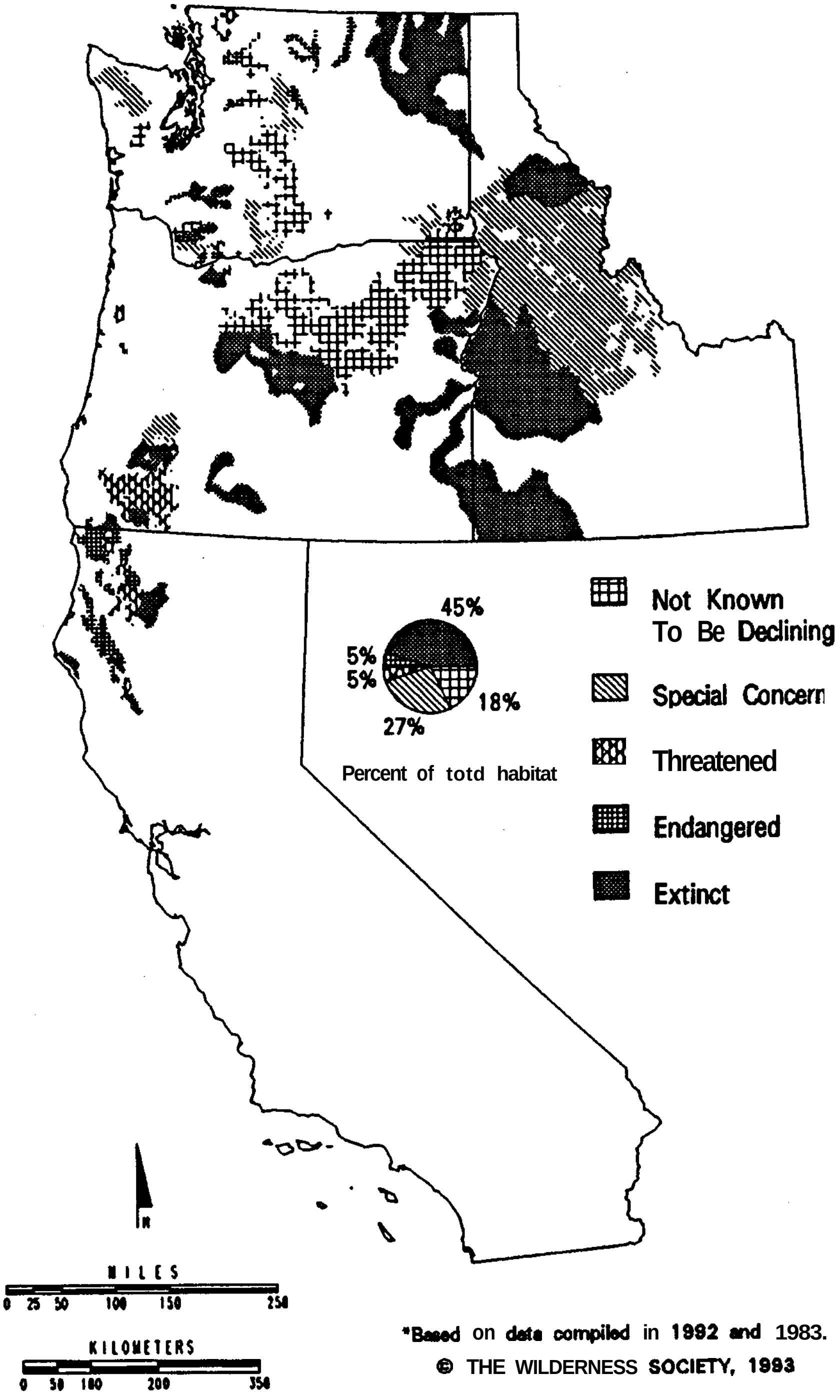
Figure 2-10. Status of winter steelhead in the Pacific Northwest and California. From TWS (1993). Reproduced with permission of the publisher.


45%
5%
5%
18%
27%
Percent of total habitat
▦ Not Known To Be Declining
▨ Special Concern
▩ Threatened
▦ Endangered
■ Extinct
N
MILES
0 25 50 100 150 250
KILOMETERS
0 50 100 200 350

*Based on data compiled in 1992 and 1983.
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Figure 2-11. Status of summer steelhead in the Pacific Northwest and California. From TWS (1993). Reproduced with permission of the publisher.

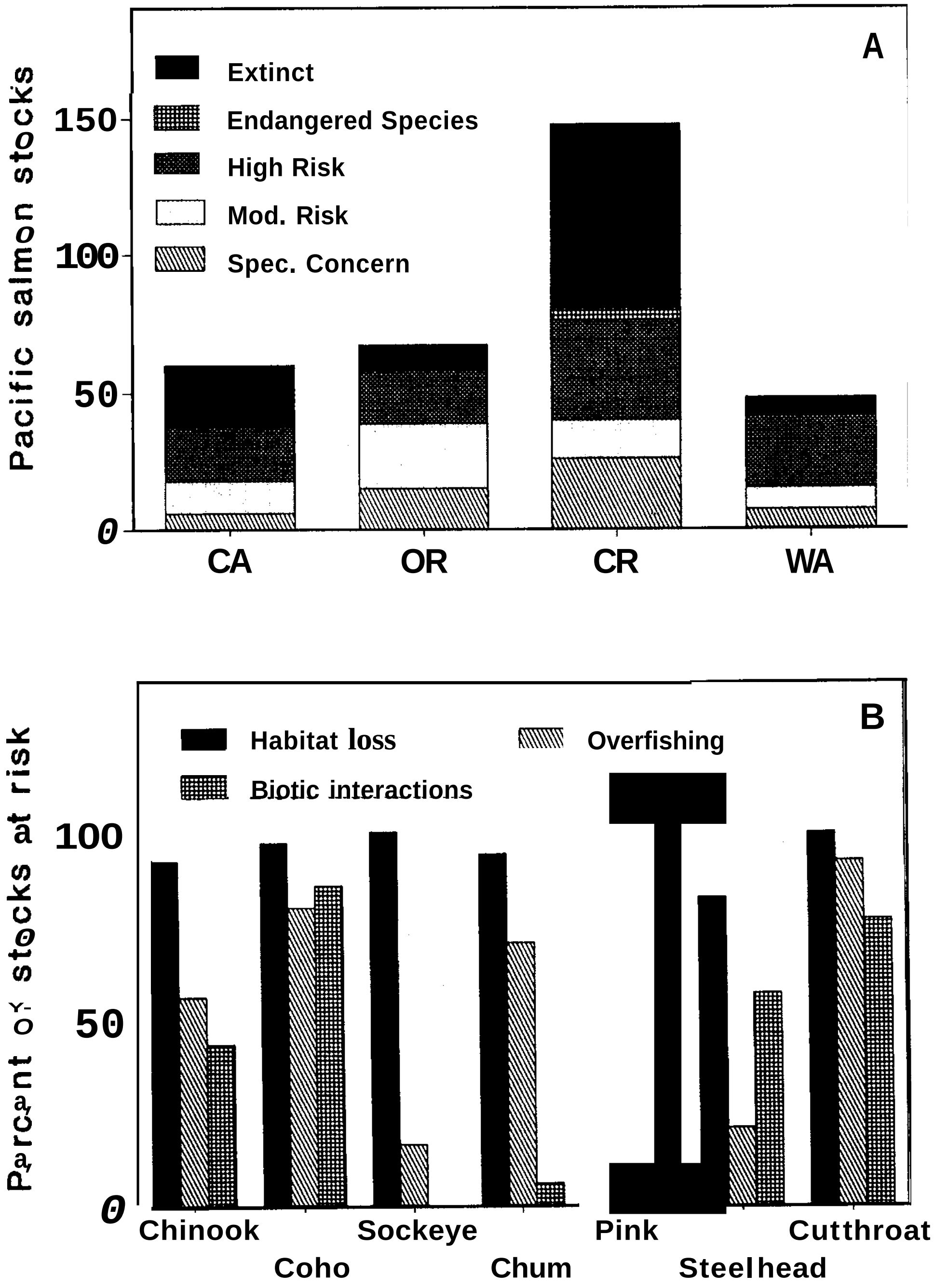


Figure 2-12. (A) Distribution of stocks of anadromous Pacific salmon *(Oncorhynchus)* in different extinction risk categories within various portions of the Pacific coast. (B) The percentage of stocks in which habitat damage, overfishing, and harmful biotic interactions have been implicated in declines of stock abundance. Figure from Bisson et al. (1992b) based on data from Nehlsen et al. (1991). Reproduced with permission of the publisher.

the Columbia Basin and along the coast (Figure 2-3). Today, coho stocks in the eastern half of their range are extinct, and stocks in the southern two-thirds of their coastal range are considered imperiled (Frissell 1993b). High numbers of threatened and endangered species in the Puget Sound and San Francisco Bay areas suggest that urbanization has contributed to the declines of native taxa.

In addition to the above reports, NMFS is now preparing status reviews of seven eastern Pacific anadromous salmonids over their ranges in the region. These reviews will incorporate information from the publications cited above as well as from state-wide status reviews prepared by the Oregon Department of Fish and Wildlife (in preparation) and the Washington Departments of Fisheries and Wildlife (WDF et al. 1993). These reviews will focus on delineating "evolutionarily significant units" (see Section 4.2.5) pursuant to potential listing as threatened or endangered under the Endangered Species Act, and they may differ somewhat from other reporting efforts.

Evidence of aquatic habitat degradation is not limited to salmonids. Counts of Pacific lamprey, *Lampetra tridentata*, at Winchester Dam on the Umpqua River in Oregon have declined from 37,000 in 1965 to 473 in 1993 (ODFW unpublished data, cited in Li et al. 1995), and lamprey returns to the Snake River numbered fewer than 20 (WDF unpublished data, cited in Li et al. 1995). Reductions in lamprey populations have likely resulted from a combination of habitat modification and the loss of salmonids as hosts; these losses demonstrate that declines are not restricted to fish species intensively harvested for consumption by humans. Amphibians, which use streams and wetlands as breeding and rearing habitats, are also highly sensitive to environmental degradation (Welsh 1990). Recent field studies in the Pacific Northwest indicate widespread declines of populations, reductions of ranges, and extinction of amphibians in forest and other ecosystems. Blaustein et al. (1994) identified habitat destruction as the major cause of amphibian losses but suggested that other factors may be important, including chemical pollution, acid precipitation, increased ultraviolet radiation, introduction of non-native species, pathogens, harvesting by humans, and natural population fluctuations.

2.3 Cumulative Effects

The widespread decline of salmonid **stocks** throughout much of the Pacific Northwest has resulted from the cumulative effects of water- and land-use practices, fish harvest, hatchery practices, and natural fluctuations in environmental conditions. The term "cumulative effects" has been used generally to describe the additive or synergistic effects of these practices on ecosystems. Another comprehensive definition of cumulative effects is provided by (Sidle **1989):** "changes to the environment caused by the interaction of natural ecosystem processes with the effects of land use, distributed through time and space, or both."

Because of the longitudinal nature of stream ecosystems, the accrual of effects is important along both spatial and temporal dimensions. Activities that take place in headwater streams influence the suitability of habitats in downstream reaches (e.g., temperature change, sediment input) and affect the response of ecosystem components to additional stresses. Similarly, activities that have occurred in the past may influence current habitat conditions through residual effects (e.g., alterations in channel morphology caused by splash dams, hydraulic mining, channelization, and revetments) and long-term, persistent effects (e.g., reduced woody debris recruitment; loss of nutrients from salmon carcasses). And finally, some activities have latent effects on aquatic systems—effects that are triggered by future environmental events (e.g., mass wasting of hill slopes, debris torrents, incision of stream channels).

In the context of conserving and restoring salmonids, the notion of cumulative effects has at least two important implications. First, individual actions that by themselves are relatively minor may be damaging when coupled with other actions that have occurred or may occur in a watershed. Historical and current patterns of land-use practices, as well as other factors, have a significant bearing on how salmonid populations will respond to further anthropogenic disturbances. Traditional management strategies that rely on site-specific analyses without regard for other activities that have occurred or are occurring within a watershed or region will generally fail to protect salmonid populations against cumulative effects. This premise underlies the development of watershed and ecosystem approaches to resource management. Second, regional declines in salmonid populations are the product of numerous incremental changes in the environment. It is thus reasonable to expect that recovery of salmonid populations will proceed in a similar fashion—through incremental improvements in habitat conditions. Few activities directed toward improving habitat are likely to have sudden and marked influences on salmonid populations, and in many cases we may be unable to detect any improvement at all amid the "noise" of natural variation in salmonid production, except over long time periods (Hall and Knight 1981). This suggests that we should temper our expectations of how rapidly ecosystem complexity and integrity can be restored (Bisson et al. 1992b). It also means that individual stakeholders can and must play an active role in salmonid habitat restoration even if tangible benefits are slow to come.

Cumulative effects of human activities have resulted in a regional landscape—including the aquatic ecosystems contained therein—that is highly fragmented with few large expanses of land (i.e., whole watersheds or basins) that are relatively intact (Doppelt et al. 1993). Early settlement of the Pacific Northwest was concentrated along low-gradient streams and rivers on relatively gentle terrain that was suitable for farming and ranching. Larger waterways served as primary travel corridors for boats as well as logs that were felled in or transported to the riparian zone and floated to downriver ports, Snagging operations removed thousands of logs annually to facilitate this river traffic (Sedell and Luchessa 1982). Similarly, roads and railroads typically were laid out in valley bottoms adjacent to rivers because gentler grades made construction easier. Diking and removing brush were commonly employed to reduce flooding of lowland areas and to allow farming and construction of houses within the historical floodplain. Streams were channelized to facilitate rapid runoff of stormwaters from watersheds.

A ***de facto*** consequence of these (and other) activities and their cumulative effects on salmonid habitats is that many of the most pristine habitats that remain are in headwater streams, where human disturbance has been less severe (Doppelt et al. 1993; Frissell et al. 1993a; Henjum et al. 1994). This situation has led to a common misperception that headwater environments are the preferred habitats of salmonids. In reality, headwater streams generally do not contain the wide array of habitats that are necessary or desirable for all life-stages of salmonids or for different fish species that have varying habitat requirements (Sheldon 1988). It is generally believed that unconstrained, aggraded floodplain reaches were once highly productive habitats for some anadromous salmonids (Stanford and Ward 1992). For example, off-channel areas adjacent to larger rivers have been shown to be important rearing habitats for salmonids during high winter flood events (Tschaplinski and Hartman 1983).

Fragmentation of habitat and the resulting isolation of populations may affect the long-term viability of salmonid stocks (see Section 4.2.4). In addressing fragmentation and connectivity of habitats for the northern spotted owl, Thomas et al. (1990) outline several general principles that are equally applicable to salmonid conservation:

- Large blocks of habitat are preferable to small blocks.
- Patches of habitat that are close together are superior to those that are far apart.
- Contiguous blocks are preferable to fragmented habitats.
- Interconnected patches are better than isolated habitat patches, and corridors linking habitats function better when they resemble the preferred habitat of the target species.

Thus, essential goals of salmonid restoration should be to prevent further fragmentation of aquatic habitats, to improve connectivity between isolated habitat patches, and to protect and restore areas surrounding critical refugia from further degradation so as to allow for the expansion of existing populations.

2.4 Strategies for Salmonid Conservation

In the last twenty years, there has been a fundamental shift away from "single-species management" of salmonids toward more holistic watershed and ecosystem approaches that seek to conserve aquatic habitats by protecting processes operating throughout the watershed. The Federal agencies responsible for administering public lands have concluded that ecosystem management is essential for arresting further habitat degradation, maintaining habitats that are relatively intact, and aiding in the recovery of at-risk species of fish (FEMAT 1993; FS and BLM 1994b, 1994c). Several recent efforts that incorporate an ecosystem perspective include the Aquatic Conservation Strategy in FEMAT (1993), the Eastside Forests Scientific Society Panel Report (Henjum et al. 1994), and the PACFISH strategy (FS and BLM 1994b, 1994c). BLM's strategy for managing wetland and riparian areas recognizes that "entire watershed condition is an important component in assessing whether a riparian-wetland area is functioning properly" (Barrett et al. 1993). EPAs Environmental Monitoring and Assessment Program (EMAP) is also based on the concept that all ecosystems existing in the landscape are integrated components and that the condition of one component affects and is affected by the condition of the others (Messer et al. 1991; Paulsen and Linthurst 1994). NMFS's coast-wide status review of coho salmon (Weitkamp et al. 1995) and steelhead trout, as well as the imminent coast-wide reviews of sea-run cutthroat trout and chinook, pink, chum, and sockeye salmon (NMFS 1994), further reflect a more comprehensive approach to resource management.

The **FEMAT** and PACFISH approaches to aquatic resource conservation as well as other published conservation strategies (Moyle and Sato 1991; Doppelt et al. 1993; Frissell et al. 1993; Henjum et al. 1994; Bradbury et al. 1995) share two common elements. First, each of these strategies recognizes the importance of identifying and protecting those habitats that retain the highest degree of integrity to serve as refugia and centers from which population expansions can occur. Second, they recognize that an effective conservation strategy must emphasize restoring ecological processes and function and must be organized at a watershed (or larger) scale. "Key Watersheds" identified by the FEMAT report, the PACFISH strategy, and the Eastside Forests Scientific Society Panel Report illustrate these concepts as applied to Federal lands.

The historical abundance of many salmonids in the Pacific Northwest was due in part to the diversity of life-history types that evolved to exploit a wide array of available habitats and that allowed temporal and spatial segregation of habitat use. In the diverse, geomorphically and tectonically unstable environments of the Pacific Northwest, well dispersed networks of locally adapted salmonids are believed to be necessary for species persistence (Frissell 1993a). This diversity enhances the ability of species to adapt to continually changing environmental conditions. Furthermore, the anadromous life-history pattern exposes these fish to a tremendous diversity of habitats, which may include small headwater streams for spawning; larger streams, lakes, or off-channel areas for rearing; still larger streams as migration corridors; and estuaries and oceans for primary growth phases. Resident trout, char, and whitefish may also spend portions of their life histories in streams and lakes of various sizes. The success of salmonid populations depends on the availability of high-quality habitats needed during each life stage.

Conservation of salmonids will require a comprehensive approach that addresses these spatial and temporal needs. Current strategies for managing Federal lands began this process, but because of the spatial distribution of Federal lands, protected watersheds presently tend to be concentrated in higher-elevation areas, forested watersheds, and headwater streams. The FEMAT report specifically cites the importance of nonfederal lands in an overall riparian conservation strategy, and Henjum et al. (1994) further stress the need to accommodate a wide variety of habitat types through the establishment of Aquatic Diversity Areas. A strategy for nonfederal lands should build upon existing conservation plans by re-establishing connectivity between habitats on Federal and nonfederal lands, and by working towards protection of habitats that are poorly represented in Federal ownership, particularly the lower-elevation streams and habitats for resident species, including nongame fishes. (Both the FEMAT and PACFISH approaches focus on anadromous salmonids.) A strategy for salmonid conservation should also provide guidance for managers so that actions at a local scale can be integrated into watershed and regional recovery plans.

Local habitat rehabilitation is essential within this broader context of conserving habitats and biodiversity across broad landscapes. Improved land-use practices and rehabilitation of riparian zones can provide many benefits, including decreased sediment transport to the stream, decreased stream temperatures, increased allochthonous nutrient inputs, increased flood-plain interaction, stabilized ground water discharge, and increased inputs of large woody debris (Naiman 1992). As natural processes and conditions are restored, downstream reaches will be improved and connections between habitats re-established (Salo and Cundy 1987), allowing greater expression of life-history diversity. Thus, private landowners can play a vital role in both improving local conditions and advancing the recovery of salmonids region wide. Furthermore, local actions can enhance other values, including water quality and quantity.

2.5 What is Ecosystem Management?

The preceding section identifies several Federal and nonfederal programs or strategies intended to foster ecosystem management as it relates to aquatic systems. A recent study by the Congressional Research Service (1994) identified no fewer than eighteen Federal agencies that have committed to principles of ecosystem management, and various state and local government and nongovernment entities have made similar commitments (Christensen et al. 1996). Yet despite the apparent widespread acceptance of ecosystem management as a paradigm, the term "ecosystem management" can be taken to mean different things by different people (GAO 1994), and some people consider the term vague or imprecise. Many definitions found in the literature have common elements, such as "sustainability" or emphasis on protection of "ecological processes or functions," but without rigorous definition, these phrases too can be considered nebulous, opening the door for misuse or misinterpretation.

The Ecological Society of America Committee on the Scientific Basis for Ecosystem Management recently reviewed and synthesized much of the existing literature related to ecosystem management (Christensen et al. 1996). They identified eight essential components of ecosystem management including 1) sustainable management of resources, 2) clearly defined and operational management goals, 3) management based on the best available science and models, 4) recognition of the complexity and interconnectedness of ecological systems, 5) recognition that ecosystems are constantly changing, 6) acknowledgement that ecosystem processes operate at multiple temporal and spatial scales, 7) the need to consider humans as integral parts of ecosystems, and 8) the importance of adaptability and accountability in management (Table 2-2). **A** key aspect of their definition is that "sustainability" is applied not to specific goods or services that ecosystems provide but rather to the ecological processes and structures that give rise to these goods or services.

We concur with Christensen et al. (1996) that these components form a sound basis for ecosystem management, and readers of ***An** Ecosystem **Approach** to **Salmonid** conservation* will find discussion of each of these elements throughout the document. Those elements pertaining primarily to ecological processes (e.g., sustainability of resources and ecosystem processes, complexity and interconnectedness of ecosystems, temporal and spatial aspects of ecological processes) are addressed mainly in **Part** I; aspects related to implementation of ecosystem management (e.g., management goals, social dimensions, adaptive management) are discussed in **Part** II. Our purpose in highlighting these elements of ecosystem management in this section is to provide a frame of reference from which to organize material presented in the remaining chapters.

Table 2-2. Essential components of ecosystem management. Based on recommendations of Ecological Society of America (1995).

Attribute	Description
Sustainability	Ecosystem management entails managing in such a way as to ensure that opportunities and resources for future generations are not diminished. Sustainability should not be evaluated based on the delivery of specific goods and services, but rather on the maintenance of the ecosystem structures and processes necessary to provide those goods and services.
Goals	Ecosystem management requires clearly defined goals. These goals should not focus exclusively on individual commodities (e.g. board feet of timber, catch of fish, visitor days). They should be explicit in terms of desired future trajectories or behaviors for components and processes necessary for sustainability.
Sound ecological models and understanding	Ecosystem management is founded on sound ecological principles, emphasizing the role of ecosystem structures and processes. It must be based on the best science and models currently available.
Complexity and connectedness	Ecosystem management recognizes that ecological processes are complex and interwoven and that this complexity and connectedness may confer particular properties (e.g., stability, resistance, resilience) to ecosystems.
Recognition of dynamic nature of ecosystems	Ecosystem management recognizes that environmental change and biological evolution are inherent properties of ecosystems and that attempts to maintain particular ecosystem "states," rather than ecological capacities, are futile over the long term in a changing environment.
Context and scale	Ecosystem management acknowledges that ecosystem processes operate over a wide range of spatial and temporal scales and that their behavior (including their response to human perturbations) at a given location is strongly influenced by the surrounding landscape or system and by the legacy of past events.
Humans as ecosystem components	Ecosystem management acknowledges that humans are components of ecosystems, as well as the source of most significant challenges to sustainability. Humans who are a part ecosystems will, of necessity, define the future of those ecosystems. Thus, ecosystem management applied alone, without consideration of social and economic systems (and their sustainability), is insufficient to ensure resource sustainability.
Adaptability and accountability	Ecosystem management recognizes that current models and paradigms of ecosystem structure and function are provisional and subject to change. Acknowledging limits to scientific understanding and adapting to new information as it becomes available are central to successful ecosystem management.

3 Physical and Chemical Processes

The freshwater and estuarine habitats of salmonids are the product of interactions among numerous physical, chemical, and biological processes (Marcus et al. 1990; Swanston 1991) operating over long- and short-term temporal scales as well as large and small spatial scales. Over millions of years, tectonic and volcanic activity in the Pacific Northwest has created a region of extreme topographic complexity, characterized by a series of mountain ranges that are oriented along a north-to-south axis and separated from one another by lowlands, plateaus, or smaller mountain ranges. Significant portions of the Pacific Northwest landscape have been reshaped by glacial advance and recession. These large-scale, long-term, geomorphic and climatic processes have created the physical template upon which rivers and estuarine systems of the Pacific Northwest have formed.

Within a watershed, topographic, geologic, and climatic characteristics control soil development and vegetation cover as well as influence the transport of water, sediments, wood, and dissolved materials from upland areas to the stream channel. These transport processes occur continuously but may be greatly accelerated during natural disturbances such as floods, debris torrents, landslides, and wildfires. The riparian zone acts as a filter that moderates the exchange of materials from terrestrial to aquatic ecosystems. In addition, riparian vegetation directly controls stream environments by providing shade and stabilizing streambanks and through the input of organic litter and large woody debris.

A useful way to conceptualize how these processes ultimately affect salmonid habitats is in terms of a hierarchy of factors (Frissell et al. 1986; Naiman et al. 1992), where each component exerts influence on other components—usually at the same or lower levels—and all components ultimately influence the character of the stream, lake, or estuary (Figure 3-1). Elements at the top of the hierarchy (e.g., climate, geology, topography, soils, and vegetation) have pervasive effects on other processes occurring in a basin or watershed (e.g., sediment delivery, hydrology, nutrient cycling,

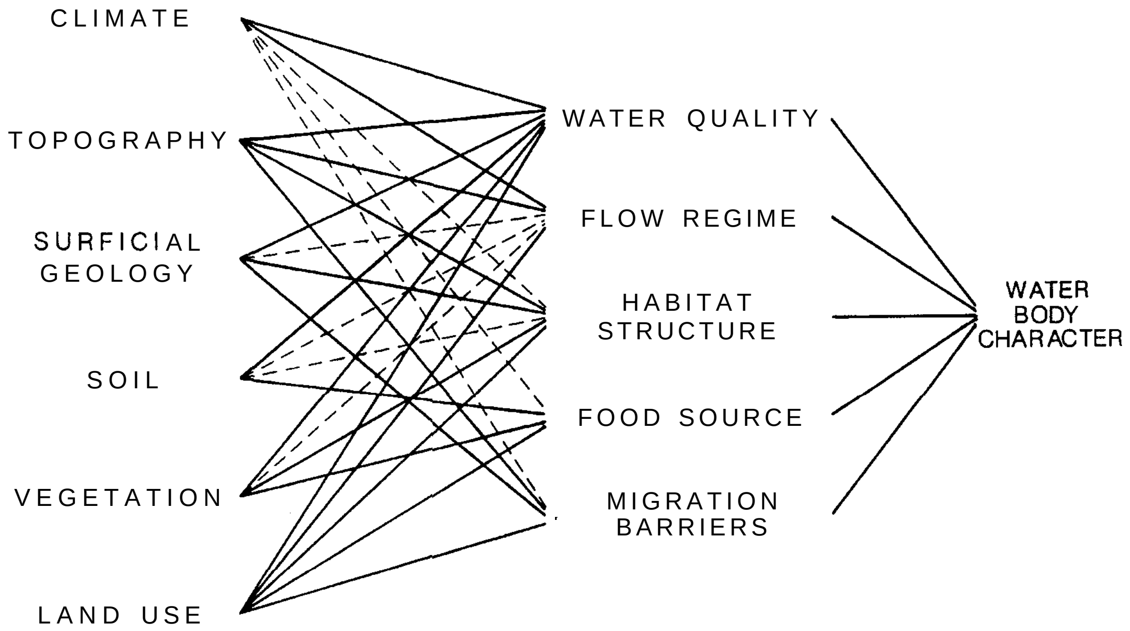


Figure 3-1. The influence of watershed characteristics on the character of aquatic ecosystems. Solid and dotted lines represent greater and lesser influences, respectively. Modified from Hughes et al. (1986). Reproduced with permission from the publisher.

riparian features) that give rise to the water body characteristics (e.g., water quality, flow regime, habitat structure, aquatic biota). Lower-tier processes generally interact with other components at the same and lower levels but may also influence components at higher levels through feedback loops. Specific characteristics of salmonid habitats are thus the manifestation of highly complex interactions among processes operating over many spatial and temporal scales.

In this chapter, we provide a broad overview of the dominant physical and chemical processes affecting the landscape and, ultimately, the aquatic ecosystems on which salmonids depend. The relative influence exerted by each specific process varies across the landscape with differences in geomorphology, geology, climate, hydrology, soil, vegetation, and other controlling factors. Consequently, the potential effects of human disturbances on aquatic systems are similarly variable in space. Our objective is to provide sufficient detail of physical and chemical processes so that regional differences in the response of ecosystems to human-caused perturbations can be understood. We begin with a review of processes that operate over large temporal and spatial scales and over which humans have minimal influence. Next we review processes that operate at smaller spatial and temporal scales and that may be substantially altered by land-use activities. Included in this discussion is a review of functional roles of riparian vegetation with respect to salmonid habitats. A summary of the effects of physical and chemical processes on salmonids and their habitats is presented in Section 3.10.

3.1 Tectonism and Volcanism

Tectonic activity operating over millions of years created the rugged montane physiography, high local relief, and steep slopes of the Pacific Northwest—structural features that control the geographic patterns of drainage systems in the region. These processes set the stage for other geomorphic processes that shape stream channels.

Direct effects of tectonics on active geomorphic processes generally are limited in spatial extent and relatively infrequent, compared to other processes discussed in this document. The Pacific Northwest is subject to large subduction zone earthquakes at intervals of severai hundred years. These large-magnitude earthquakes may cause subsidence in soft alluvial and coastal fills, creating zones of deposition (Atwater 1987; Darienzo and Peterson 1990), and they may also trigger mass movements of soil.

Volcanic activity has been less significant regionally than tectonics and glacial processes, but at local sites it has resulted in catastrophic readjustments of the landscape. Geomorphic impacts depend on the geochemical type of volcanism. Explosive eruptions of silicic volcanoes directly reshape the landscape, blocking and diverting drainage systems by ash flows, filling valleys or channels with mudflows, and causing major inputs of sand and silt-sized sediments from tephra (airborne ash). Basaltic volcanic centers may block and divert drainage systems through lava flows and cinder eruptions and also release limited amounts of tephra. Recently active silicic volcanic centers are limited to the Cascade Range (Sarna-Wojcicki et al. 1983). Basaltic volcanic centers active in the Holocene are found in the Cascades from southern Washington to California, a few areas of eastern Oregon, and in the eastern Snake River Plain of Idaho.

Volcanic mud- and ashflows commonly occur from volcanic eruptions in Cascade Range volcanoes. Mudflows have produced the most widespread geomorphic effects of past eruptions and can inundate valley floors with deposits less than one meter to tens of meters thick. Mudflows caused by the 1980 eruption of Mount St. Helens inundated valleys and completely buried pre-existing river channels. Channels subsequently re-established on the mudflow deposits through alternating episodes of incision, channel widening, and aggradation over a period of at least several years (Meyer and Martinson 1989). During the adjustment period, sediment yields were much higher than before the eruption. Geomorphologic adjustments have been prolonged by landslides on slopes that were destabilized by the eruption. Ash flows also move down valleys and bury valley floors (Crandell 1976), while tephra may be deposited many kilometers from the source.

3.2 Glaciation

The landscape of the Pacific Northwest has developed under alternating glacial and interglacial periods over the last one million years or longer. Glaciation has affected the region's landscapes through 1) direct modification of mountain areas and limited lowland areas by glaciers; *2)* eustatic sea-level lowering, which has had major effects on coastal rivers and estuaries; **3)** glacial-interglacial climatic changes that have influenced the hydrologic regime; and **4)** climate-driven changes in vegetation cover that have affected hillslope and stream processes (Table **3-1).** In general terms, glacial periods are times of rapid sediment transfer from uplands to lowlands and to the ocean by glacial advance and meltwater transport in glaciated areas and by increased streamflow in unglaciated areas. Interglacial periods tend to be periods of sediment accumulation in upland valleys with limited fluvial transfer out of the uplands (Thorson 1987).

Table 3-1. Past controls and effects on landscape development in the Pacific Northwest.

Period	Controls	Probable geomorphic and ecological effects
Glacial	In glaciated and periglacial areas:	
	Advance of Cordilleran ice sheet; development of mountain ice sheets and alpine glaciers; very cold climate with reduced precipitation.	Glacial erosion and deposition and formation of outwash trains in valleys; periglacial churning and mass movement, intensified mechanical weathering; glacial meltwater discharge; displacement of interglacial ecological communities; vegetation cover absent or greatly reduced.
	In unglaciated areas:	
	Lowered sea level; cold climate with reduced precipitation.	Displacement and shrinkage of estuary areas; reduced vegetation cover; mechanical weathering, mass movement and slope erosion rates greater than modern; increased streamflow and fluvial sediment transport; accumulation of coarse valley fills; reduced organic inputs to streams.
Late Glacial and early Holocene	Retreat and downwasting of glaciers; rapid sea-level rise; warming; effective moisture greater than modern ca. 14,000 to 11,000 years ago, then less than modern until ca. 7,000 years ago.	Glacial deposition and exposure of glaciated land surfaces; landward displacement of estuaries, increase in estuary depth and area; mass movement and slope erosion rates decreasing but still greater than modern; streamflows probably greater than modern; stabilization and then incision of valley fills; increasing vegetation cover and changes in community composition; increased organic inputs to streams, but still less than modern; minor fluctuations in alpine glaciers.
Middle Holocene to modern	Sea-level stable; climate approaching modern conditions, with short-term fluctuations.	Estuaries filling and shallows developing; slope stabilization and decrease in mass movement rates; decreased mechanical and increased chemical weathering; streamflows near modern, with short-term fluctuations; continued but slowed incision of valley fills; development of modern ecological communities; high rates of organic inputs to streams; minor fluctuations in alpine glaciers.

Where residence times of sediment accumulations or recurrence intervals of events are thousands of years (Dietrich et al. 1982; Kelsey 1982), glacial-interglacial transitions may be the most important periods in landscape formation.

During the last glacial period, about 22,000 to 15,000 years ago, ice sheets and mountain glaciers were developed in many areas of the Pacific Northwest; sea level was about 100 m below present, exposing large areas of the continental shelf. The Cordilleran ice sheet extended south from British Columbia, covering the Puget Lowland, northern Cascades, Okanogan Valley, and upper Columbia Valley in Washington. South of the ice sheet, mountain ice sheets and glaciers were widely distributed in the mountainous regions of Washington, Oregon, Idaho, and the Siskiyous of Northern California (Crandell 1965; Porter et al. 1982). Climate of the glacial period was much colder than today. Although effective moisture in the Pacific Northwest was less (Thompson et al. 1993), runoff likely was as high or higher than today, because of changed land-surface conditions. Down the valley from glaciers and in unglaciated watersheds, frost weathering and mass wasting were probably more intense than at present. River systems probably had greater streamflow and transported greater sediment loads. In addition, enormous ice jams periodically developed and broke, resulting in catastrophic floods that formed the coulees of eastern Washington and deposited the deep soils of the Willamette Valley.

In addition to these physical changes, ecological changes resulting from glacial climates may have also influenced geomorphic processes. Preglacial ecosystems of these areas were significantly displaced by glaciation; species and stocks present today in these ecosystems likely existed in refugia south of or at lower elevations than the glaciers. In unglaciated parts of western Oregon and Washington, the vegetation consisted of tundra close to glaciers and subalpine parkland elsewhere, including the Oregon Coast Range (Worona and Whitlock 1995). In eastern Washington, a sparse periglacial steppe was present (Barnosky et al. 1987; Whitlock 1992; Thompson et al. 1993). In the more sparsely vegetated landscape of the last glacial period, less large organic debris was available to influence streams and valley floors. Present environmental conditions have prevailed in this region for the last 6,000 to 8,000 years. Current conifer forest communities did not become established until 5,000 to 2,000 years ago (Whitlock 1992; Worona and Whitlock 1995), and in glaciated watersheds of western Washington, stream channels reached conditions similar to those of the present by about 6,000 to 8,000 years ago (Benda et al. 1994). As density and height of forest stands increased with climatic amelioration, woody debris exerted a stronger influence on stream and valley morphology. Some channel incision and narrowing of meander belts probably continued into the late Holocene.

A general model of river channels based on empirical evidence from several parts of the world suggests that channel changes from glacial to interglacial periods follow a specific sequence. Braided channels dominate during glacial periods. During interglacial periods these change to transitional, braided, meandering channels with mid-channel bars but well-defined thalwegs, and then to large meandering channels adjusted to higher-than-present discharge. Finally, smaller meandering channels develop during stable conditions typical of post-glacial periods (e.g., late Holocene; Schumm and Brakenridge 1987). In the Pacific Northwest, the late-glacial to early Holocene period was likely characterized by channel incision into thick glacial-period valley fills, formation of terraces, sediment yields higher than present as rivers downcut, and significant changes in channel morphology because of changed hydrologic and sediment regimes (Benda et al. 1994).

In addition to the changes in inland watersheds described above, coastal rivers were directly affected by lowered sea level during glacial periods (McDowell 1987). At the last glacial maximum, global sea level was 100 m or more below the present sea level, and the shore was 10 km or more west of its present location. Coastal streams flowed across the exposed continental shelf, perhaps in incised valleys. Estuaries were very limited in extent. As global deglaciation began, sea level initially rose very rapidly creating deep coastal estuaries. Beginning 10,000 years ago, the rising sea level continued at a decreasing rate, and it has fluctuated close to the present level since 4,000 years ago. Shallow-water conditions in estuaries, including mud and sand flats, have become established only recently (McDowell 1986, 1987).

3.3 Wildfires

The historical frequency of fires varies over the landscape as a function of climate and vegetation type. Fires in high-elevation communities of subalpine fir, western hemlock/red cedar, lodgepole pine, and grand fir tend to recur at an interval of decades to centuries; low- to mid-elevation juniper, ponderosa pine, Douglas-fir, and white fir forests typically experience fires at intervals of several years to a few decades. Little information is available about the historical frequency of fire in grassland, shrubland, and woodland communities east of the Cascade Crest (Agee 1994). Plant assemblages of stiff sagebrush and Sandberg's bluegrass may have biomass sufficiently low to prevent large-scale fires, while other communities, including various fescue and bluebunch wheatgrass assemblages, may have sufficient biomass to carry fire but lack sources of ignition (i.e., lightning) during the periods when they are most combustible (Agee 1994). Fire frequency in other sagebrush and woodland communities is poorly documented. In the Cascade region, wildfire regimes are highly variable. Morrison and Swanson (1990) reconstructed fire histories at two locations in the central and western Cascades and estimated recurrence intervals of approximately 95 and 149 years, respectively (range 20–400 years); most fire-created patches were less than 10 hectares in size. In the Coast Range, higher humidity, more lush vegetation, and less frequent lightning storms combine to reduce the frequency of wildfire; however, under dry summertime conditions, the effects of wildfire in dense timber stands can be substantial. During the period 1933–1951, four fires in the Coast Range of Oregon, collectively known as the Tillamook fires, burned more than 260,324 hectares (643,000 acres) and had significant and long-lasting effects on forest and riparian communities. Although these fires were human-caused, they demonstrate the potential for forests in the Coast Range to burn under certain circumstances.

Riparian areas generally are characterized by a higher percentage of deciduous plants than is found in surrounding uplands. In addition, local microclimates tend to be cooler, resulting in moist soils and high fuel moisture, especially in floodplain woodlands. Because of these attributes, riparian areas do not burn, or they burn at lower intensity than forests in upland areas. As such, they may buffer aquatic communities from some of the effects of wildfire. However, in headwater reaches and at higher elevations, stronger winds and greater biomass may facilitate fires of relatively high intensity. Consequently, it is difficult to generalize about the effects of fires on the riparian zone (Agee 1994).

Fires in upland areas and riparian zones can affect aquatic ecosystems by altering vegetation cover, which in turn influences erosion and sediment transport, water infiltration and routing, the quantity of nutrients reaching streams, the amount of shading, and the input of large woody debris into the system (Wissmar et al. 1994). The extent of impacts is generally related to the intensity of the burn. In high intensity fires, soil organic matter that helps hold soils together is consumed, increasing the susceptibility of soils to erosive forces. In addition, volatilization of certain compounds can cause the surface soil layer to become hydrophobic, thereby reducing infiltration of water and increasing surface runoff (Marcus et al. 1990). The combined effects of vegetation loss and hydrologic changes can alter the frequency of severe debris torrents (Wissmar et al. 1994). Nutrients such as phosphorous, nitrogen, and sulfur may be volatilized into the atmosphere (Everest and Harr 1982) or lost through leaching and soil erosion. The loss of riparian vegetation can increase exposure to solar radiation, causing streams to warm. Inputs of large woody debris may also change following fire in the riparian zone. In speculating about the effects of the Yellowstone fire of 1988, Minshall et al. (1989) hypothesize that large woody debris in streams would likely increase immediately following the fire—from augmentation of existing woody debris with falling branches—then decrease through time because new growth contributes little to instream woody debris.

Humans have significantly altered natural fire regimes through land-use practices and an extensive and long-term focus on fire suppression. As a result, significant changes in forest vegetation have resulted. East of the Cascades, fire suppression has led to shifts in vegetation from historically open stands of ponderosa pines and western larch to stands with dense understories of Douglas-fir and grand fir (Mutch et al. 1993). Ponderosa pines are well adapted to frequent, low-intensity burns that were characteristic of eastside forests. These fires tended to prevent fire intolerant species from invading. Drought and subsequent insect infestations have killed many understory trees, allowing fuels to accumulate and increasing the probability of high intensity fires (Wissmar et al. 1994). Consequently, ecosystems that once experienced frequent but small wildfire disturbances are now prone to infrequent but much more catastrophic events.

3.4 Sediment Transport

Sediment transported from upland areas into stream channels determines the nature and quality of salmonid habitat in streams, rivers, and estuaries. The development and persistence of morphological structures used for spawning, incubation, and rearing depend on the rate at which sediment is delivered and the composition of deposited materials. Sediment delivery rates and composition, in turn, are controlled by climate, topography, geology, vegetation, and hydrology. Local variation in these watershed characteristics ultimately determine the type and quality of habitat found in a given system.

Land-use practices, through alteration of soil structure, vegetation, and hydrology, can significantly alter the delivery of fine and coarse sediments to streams, thereby affecting salmonid habitats. In this section, adapted primarily from Swanston (1991), we discuss surface erosion and mass wasting, the dominant forms of sediment transport, as well as environmental factors that influence these processes. The routing of sediments within the stream channel and the role of large woody debris in controlling sediment movement are discussed in Sections 3.5 and 3.9.5 of this document.

3.4.1 Surface Erosion

Surface erosion results from rain and overland runoff. Particulate and aggregate materials are relocated via a two-step process: detachment then downslope transport of detached materials. Detachment is influenced by the size and compaction of particles and by the protective cover of organic litter and plants. Slope gradient and length, rainfall intensity, and soil infiltration rate determine transport rate (Swanston 1991). Initiation of surface erosion may be caused by landslides, fire, logging, rain, drop splash from forest overstory, animal activity, freeze-thaw phenomena, or any other surface disturbance of soil. Surface erosion rarely occurs on undisturbed forest lands west of the Cascade crest because of high infiltration rates, though it may occur in areas with steep (> 27°) slope gradients (Swanson et al. 1987). In sparsely vegetated lands east of the Cascades, the potential for surface erosion is greater because of the lack of groundcover.

Most surface sediments that reach stream channels result from channelized erosion (rilling and gullying) and sheet erosion (Brown 1980; Swanston 1991). Channelized erosion occurs when flows are concentrated and restricted by landforms, usually following heavy storms or snowmelt (Beschta et al.

1995). It is considered the most significant form of surface erosion on forest lands (Brown 1980). Although uncommon in undisturbed forested situations, rills may occur when infiltration capacity is reduced (Megahan 1991). In contrast, nonchannelized erosion develops from detachment begun by raindrop-splash and overland flow (sheet erosion) or by gravitational and wind movement of dry particles (dry ravel). These processes generally occur on exposed soils and tend to remove soil uniformly over an exposed area. Sheet erosion tends to be of greater significance on low-gradient agricultural lands than on forested lands, whereas dry ravel occurs on steep slopes in soils lacking cohesion (Swanson et al. 1987; MacDonald and Ritland 1989).

3.4.2 Mass Wasting

Mass wasting—slumps, earthflows, landslides (or debris avalanches), and soil creep—is often a major component of sediment delivery to streams, particularly in mountainous regions where surface erosion is minor (Swanson and Dyrness 1975). Generally episodic in nature, mass wasting can provide large quantities of sediment and organic material to streams. Reeves et al. (1995) suggest that historically in the Coast Range, these periodic natural disturbances (sometimes associated with wildfire) served to replenish large woody and coarse sediment in streams at intervals ranging from decades to several centuries or more. Following these disturbances, natural erosion and aggradation processes gradually modified these disturbed reaches, causing a succession of different habitat conditions for salmonids. This variation in space and time created areas of naturally excellent and poor salmonid habitat. By increasing the frequency (both spatial and temporal) and altering the nature of these disturbances (e.g., reducing the quantity of large woody debris associated with mass failures), humans have degraded and simplified stream habitats.

Slumps and earthflows generally develop in deeply weathered soils. These often occur in sedimentary geology (siltstones, sandstones, mudstones) and volcaniclastic rocks, In soils with primarily clay-sized particles, low soil permeability restricts groundwater movement and causes puddling and fluid soils (Swanston 1991). These unstable soils produce slumps and earthflows. Slumps are the sliding of soil blocks along a concave surface, and earthflows often begin as slumps or a series of slumps. Once initiated, rheological flow of the clay fraction keeps the individual soil blocks moving downslope like a viscous fluid in earthflows. Earthflows tend to be seasonal with most movement occurring after heavy rains have saturated soils. These flows are slow moving, ranging from 2.5–2,720 $cm \cdot yr^{-1}$ (Swanston 1991) and may eventually protrude into the stream channel, where they are gradually eroded away. As they erode, residual lag deposits may form, which can increase channel gradient downstream through the accumulation zone. These areas, if in otherwise "sediment poor" reaches, and if they contain coarse sediments, may increase the habitat diversity in a morphologically uniform channel and have a long-term beneficial effect on fish habitat.

Soil creep is soil movement that is imperceptible except by measurements taken over long periods of time. Carson and Kirkby (1972) identify causes including reworking of the surface soil layers because of frost heaving, steady application of downward sheer stress, and random movements from organisms or microseisms. Continuous creep tends to occur in clay soils and is absent in coarse-grained soils.

Landslides typically occur in shallow noncohesive soils on steep slopes overlying less permeable bedrock (Beschta et al. 1995). Conditions causing landslides include 1) zones of weakness in soil or bedrock, 2) wind stress transferred to the soil by trees, 3) deformation caused by soil creep, 4) drag caused by seepage pressure, and 5) removal of slope support by undercutting. Landslides—relatively dry soil masses—are distinguished from debris flows, which are typically saturated. When landslides enter stream channels during floods, they become debris flows—large volumes of water containing soil, rock, and, frequently, large organic debris. These flows scour the channel and severely modify fish habitat as they move rapidly downstream. As debris flows move downstream into higher order channels, their effects become less pronounced because of increasing streamflow.

3.4.3 Factors Affecting Erosion and Sedimentation Rates

The magnitude, locations, and frequency of sediment delivery to active channels is highly dependent upon climate, local topography, soil type, soil saturation, vegetative cover, organic matter, depth and degree of weathering, and degree of upslope disturbance (Swanston 1991; Beschta et al. 1995; OWRRI 1995). Rain-dominated watersheds tend to yield more sediment than snow-dominated systems, although interbasin variability is quite high. Larson and Sidle (1981) examined data from 13 relatively undisturbed watersheds and reported sediment yields of 2.0 to 40.7 $tonnes \cdot km^{-2} \cdot yr^{-1}$ for rain-dominated systems. For snow-dominated systems, sediment yield typically ranged from 1.6 to 6.1 $tonnes \cdot km^{-2} \cdot yr^{-1}$; however, two watersheds had substantially higher yields of 39.9 and 117.1 $tonnes \cdot km^{-2} \cdot yr^{-1}$ (see Swanston 1991). Within-year variation in sediment production can also be high. Larson and Sidle (1981) reported differences in sediment yield among years of an order of magnitude or more for both rain-dominated and snow-dominated systems.

The timing, frequency, and type of precipitation influences the rate and yield of sediment delivered to stream channels. In rain-dominated regions, sedimentation and allochthonous inputs are minimized during summer low-flow periods. Sedimentation increases during the wet months of September to February when soils are saturated and landslide hazards are highest. In snow-dominated regions, sedimentation is greatest during periods of rapid snowmelt or during high-intensity rain storms, when high streamflows occur and entire hillslope and channel systems erode (Swanston 1991).

Topography influences slope steepness, length, elevation, and aspect. Runoff energy is highest on steeper slopes with greater slope length, which increase the volume and velocity of water moving downslope. Failures that occur on lower areas of the hillside nearer streams have a greater potential of reaching the stream.

Parent material and soil types also determine soil texture and erodibility. Erodible soils include those derived from granite, quartz diorite, granodiorite, Cenozoic nonmarine sediments, and schist (Beschta et al. 1995). Diorite and various metamorphic, rock-derived soils have intermediate erodibility, and nonerodible materials include andesite, basalt, peridotite, serpentinite, and pre-Cenozoic and Cenozoic marine sediments. Important soil properties affecting mass wasting include cohesion, structure, porosity, moisture capacity, drainage, chemical properties, and soil depth, all of which are affected by the relative proportion of clay, silt, and sand in the soil (Swanston et al. 1980). Typically, soils with little cohesion, structure, or porosity, low moisture capacity, and poor drainage are more likely to erode.

Vegetative cover tends to reduce sediment transport by reducing detachment rate and through the binding capacity of root masses (Larson and Sidle 1981; Harvey et al. 1994). Organic matter, utilizing water as the cementing agent, helps to form aggregates that tend to be more resistant to detachment and transport (Dyrness 1967).

3.4.4 Regional Differences

East of the Cascades, soils are most susceptible to surface erosion, but mass wasting events can be locally important. Slumps originate in fine textured soils, while debris-torrent failures occur in weakly cohesive ash Harvey et al. (1994) suggest that the high infiltration rates in most soils of the inland Pacific Northwest region make them less susceptible to surface erosion unless slopes are greater than 30% and not vegetated. Compacted ash and pumice soils on shallow slopes are susceptible to gullying because of their low density and cohesion. In areas characterized by coarse, cohesionless soils and periods of drought, dry creep and sliding of materials from denuded slopes may be an important source of local surface erosion (Swanston 1991). Dry ravel is significant on slope gradients greater than 22" in pumiceous, cindery, and ashy soils—conditions found in specific areas in the central Oregon plateau and eastside portions of the Cascade Range in Washington. Dry ravel is also common in dryer parts of Idaho, southwestern Oregon, and the Cascades in Oregon (Swanson et al. 1987).

Mass wasting occurs with high frequency in the western Cascade Mountains and Coast Range (MacDonald and Ritland 1989; Beschta et al. 1995). Wet climatic conditions in the Coast Range and valleys tend to promote deep soil formation and clays, which are prone to slow continuous failures, including slumps, soil creep and earthflows. At higher elevations near the Cascade crest, shallow, cohesionless soils overlying slightly weathered bedrock are susceptible to landslides. Sediment budgets from three sites illustrate these regional differences. The wet, snowmelt-dominated, glaciated, and tectonically active Queen Charlotte Islands of British Columbia have sediment yields an order of magnitude greater than drier, snowmelt- dominated, granitic lands of central Idaho, while in the rain-dominated regions of western Oregon and Washington, yields are intermediate to the other two regions (MacDonald and Ritland 1989).

3.5 Channel Morphological Features and Their Formation

Stream conditions important for aquatic habitat can be observed over a range of scales from an entire drainage network to a reach to a channel unit (Gregory et al. 1991). Average values of many stream characteristics, such as width, depth, velocity, and bed material size, vary systematically in a downstream direction, There are, however, important patterns of variation at local scales, such as the reach and the channel unit scales. Reaches are stream and valley segments, typically 1–10 km long, within which gradient, valley width, and channel morphology are relatively homogeneous and distinct from adjacent segments. Reach-scale variation is controlled by geologic factors such as rock type, geologic structure, and location of geomorphic features such as terraces, alluvial fans, and landslides (Table *3-2)*. In many streams, high-gradient reaches with narrow, constrained valley floors are interspersed with lower-gradient, alluvial reaches with wide valley floors (Grant et al. 1994).

Table 3-2. Reach classes in small Oregon streams. From Frissell et al. 1986. Reprinted with permission from the publisher.

Gross typology	Morphogenetic class*	Morphogenetic process	Relative length	Mean slope†	Dominant substrates	Developmental trend	Potential persistence‡
EROSIONAL	Bedrock outcrop	Irregular bedrock; resistance to weathering	Moderate to short	Variable: moderate to steep	Bedrock	Stable; all sediments transported	Long term
(Zones of exposure of bedrock floor or trend toward degradation of bed)	Colluvium (nickpoint)	Downcutting through landslide or torrent debris	Moderate to short	Steep, later becoming moderate	Boulders, cobbles, clay soil	Active degradation (unless reloaded	Generally moderate; depends on deposit size
	Torrent scour	Channel scour by debris torrent or flood	Moderate to long	Moderate to steep	Bedrock, some boulders	Transport of most sediments; local aggradation	Moderate (due likely to recruitment of constructional features
Channel pattern: straight	Alluvium	Downcutting through alluvium of old constructional reach	Moderate	Moderate	Cobbles, gravels	Slow degradation	Moderate to short term
	Root blockage	Channel shift after colluvium or debris jam blockage; tree roots delay downcutting	Short to moderate	Moderate to low	Tree roots, gravels, cobbles, clay soil	Stable period followed by degradation	Short term; very short if small roots
CONSTRUCTIONAL	Bedrock outcrop	Sediment storage behind resistant bedrock features	Variable	Low	Gravels, fines, bedrock	Stable; inputs balance outputs	Long term
(Zones of aggradation and alluvium)	Colluvium	Sediment storage behind landslide or debris torrent deposits	Variable	Low	Gravels, cobbles, fines	Degradation, shortening (unless reloaded)	Long term to moderate (depends on deposit size)
Channel pattern: straight often verging on braided	Large woody debris	Sediment storage behind large logs or debris jams	Moderate	Low	Gravels, fines, wood	Net aggradation until decay or washout	Moderate, sometimes long term
	Small woody debris	Sediment storage behind jam of small debris	Short	Low to moderate	Gravels, cobbles, fines, wood	Aggradation, then quick washout	Short term

* Morphogenetic classes are further subdivided by segment class, whether banks are clayey colluvium or gravelly alluvium, whether sideslopes allow lateral migration, and by riparian vegetation state.

† Slope scale: moderate = same as segment slope, low = less than segment slope, and steep = greater than segment slope.

‡ Persistance scale: long term = > 100 years, moderate = 20–100 years, and short term = < 20 years.

In such streams, width/depth ratios of a channel, channel bed morphology, and relationships of a channel to the valley floor will vary significantly from reach to reach. Reach-scale variations influence the location of spawning areas and types of fish that inhabit a stream system (Grant et al. **1994;** Montgomery **1994).** Consequently, reach-scale variations are an important consideration in watershed planning.

Channel units or habitat units consist of morphological features such **as** pools, glides, riffles, rapids, stepped-pool sequences, cascades, and steps (Table 3-3). Channel units exert an important influence on local flow hydraulics and bed-sediment characteristics (Grant et al. **1990).** Channel units, therefore, provide the local habitat context for aquatic insects, fish, and other animals that inhabit stream channels.

Table 3-3. Types of channel (habitat) units. From Grant et al. (1990). FS (1993), and Beschta and Platts (1986).

Type	Morphology	Hydraulic characteristics	Ecoloaical function
Pool	Deepest, lowest gradient unit; depth varies within unit: may have asymmetrical cross-section; may accumulate fine bed material at low flows.	Slow, tranquil, sub-critical flow without hydraulic jumps during low flow; scour, turbulence, and energy dissipation during high flow.	Fish rearing; invertebrate production.
Glide (run)	Intermediate, uniform depth; symmetrical cross-section; gravel or cobble-bedded.	Tranquil subcritical flow generally without hydraulic jumps.	Fish rearing; invertebrate production.
Riffle	Shallow depth; gravel or cobble bedded.	Tranquil, generally subcritical flow with small hydraulic jumps over boulders or cobbles.	Invertebrate production: salmon and trout spawning; steelhead rearing; may be winter cover for salmon and trout; aeration.
Rapid	Shallow depth; often have transverse ribs of emergent boulders and pocket pools; common emergent boulders.	Between 15%–50% of area in supercritical flow (jumps, standing waves) at low flow.	Aeration; summer cover for salmonids.
Cascade	Shallow depth; steeper overall than rapid; consists of a series of short steps over boulders or bedrock ledges; common emergent boulders.	Greater than 50% of area in supercritical flow at low flow.	Aeration; may be migration barrier (if large).
Step	Isolated small falls, 1–2 m high and less than one channel width in length over boulders, bedrock or large woody debris; common emergent boulders, bedrock or wood; steepest and shallowest units.		Aeration; may be migration barrier (if large).

The relative abundance of different channel unit types, such as pools or cascades, varies from reach to reach in response to variation in controls such as bedrock type, reach gradient, mass movement features, sediment size, and position in the channel network. Steep reaches, associated with resistant bedrock types or with coarse substrate deposits that intersect the channel, are dominated by cascades, rapids, or steps with limited pool, glide, and riffle area (Grant et al. 1990). Overall, these channel units combine to form a step-pool channel morphology in steep reaches. The steep channel units are associated with boulder-sized bed material. Biological processes also play a role by creating steps and pools adjacent to accumulations of large woody debris (Beschta and Platts 1986). Although formed of boulder-size material that exceeds the competence of most high-flow events (i.e., mean annual floods), stepped-bed channels in steep mountain streams of the Pacific Northwest are not residual features but are in equilibrium with the modern hydrologic regime (Grant et al. 1990). The channel units are reworked by flow events with recurrence intervals of 25–50 years. In less steep reaches, the cascades, rapids and steps are less frequent; bed material is dominated by cobbles and gravels rather than boulders; and the abundance of pools, riffles and glides is higher. Active bedload, transported during frequent high flows (mean annual flood), accumulates in bars that are positive relief features on the channel bed. These bars result in pool-riffle channel morphology that is expressed at moderate to low flows. Pools are located at points of scour during high flow events, and riffles are formed by bar fronts (Lisle 1982; Beschta and Platts 1986; Wohl et al. 1993).

Human modification of flow and sediment regimes can modify the abundance and character of channel units. Human impacts resulting in net aggradation, for example, tend to reduce pool area and depth (Lisle 1982; Beschta and Platts 1986). Human impacts that decrease woody debris input to the channel can have the same effect, as can flood-induced aggradation (Lisle 1982). Human impacts that result in net degradation may also reduce pool area if bedload is depleted and bedrock is exposed in the channel bed.

3.6 Hydrology

The flow in streams and rivers represents the integration of the climate, topography, geology, geomorphology, and vegetative characteristics of a watershed. Precipitation may be intercepted by the vegetation and subsequently evaporate, or it may reach the ground either directly or as throughfall. Water reaching the ground either evaporates, infiltrates into the soil, or flows overland until it reaches the stream or an area where infiltration is possible. Water that infiltrates the soil may be taken up by plants and transpired back into the atmosphere, remain in the soil as stored moisture, percolate through the soil into deep aquifers, or enter streams via subsurface flow. Each of these processes affects the amount and timing of streamflow.

Land-use disrupts natural hydrologic processes, altering the amount of evaporation, transpiration, and runoff, the routing of water through the system, and the temporal patterns of streamflow. Regional differences in the hydrologic cycle can affect the response of a watershed to human disturbance. Consequently, an understanding of basic hydrologic processes is critical to understanding how land-use practices influence streamflow and how these effects vary across the landscape. This section provides a brief overview of hydrologic processes that occur in a watershed, with emphasis on those processes that may be substantially modified by human disturbance. A thorough review of hydrologic processes can be found in Swanston (1991).

3.6.1 Precipitation

The amount, form, and timing of precipitation differs dramatically across the Pacific Northwest, with the primary controlling factors being latitude, elevation, and proximity to the ocean and mountain ranges (Jackson 1993). Moisture-laden air generated over the Pacific Ocean is uplifted and cooled as it approaches mountainous regions, causing water to condense and fall as precipitation. After the air mass passes over these mountains it warms again, increasing its capacity to hold moisture. Thus, areas on the east slope of mountain ranges receive less rainfall than western slopes of comparable elevation, the so-called "rain shadow" effect. Convection storms (i.e., storms generated by heating and upward expansion of air masses near the earth's surface) may also be a significant source of precipitation during the spring and summer months in mountainous regions and continental climates east of the Cascade and Sierra Nevada crests. These storms tend to be localized events of high intensity and relatively short duration.

Three general precipitation systems are in the Pacific Northwest: rain-dominated, transient-snow, and snow-dominated systems (Table 3-4). Rain-dominated systems include coastal mountains, lowland valleys, and lower elevations of the Cascade and Sierra Nevada ranges, characterized by moderate to high precipitation that falls primarily as rain from late fall to early spring. In some coastal regions and lower elevations of the western Cascades, fog drip from forest canopies may also constitute a significant part of the total precipitation (Oberlander 1956; Azevedo and Morgan 1974; Harr 1982). The transient-snow zone includes mid-elevation areas of the Cascades, northern Sierra Nevada, and Olympic

Table 3-4. Precipitation patterns for selected ecoregions in the range of anadromous Pacific salmonids. Data from Omernik and Gallant (1986).

Ecoregion	Mean annual precipitation cm (inches)	Dominant form	Season of greatest precipitation
Coast Range	140 – 318 (55 – 125)	Rain.	Mid fall – early spring.
Puget Lowlands	89 – 127 (35 – 50)	Rain.	Mid fall – early spring.
Willamette Valley	89 – 114 (35 – 45)	Rain.	Mid fall – early spring.
Central CA Valley	38 – 64 (15 – 25)	Rain.	Winter.
Southern & Central CA Plains & Hills	51 – 102 (20 – 40)	Rain.	Winter.
Cascades	127 – 254 (50 – 100)	Rain (low elevation); snow (high elevation).	Mid fall – early spring.
Sierra Nevada	46 – 216 (18 – 85)	Rain (low elevation); snow (high elevation).	Mid fall – early spring.
Eastern Cascades Slopes & Foothills	30 – 64 (12 – 25)	Snow.	Mid fall – early spring.
Columbia Basin	23 – 64 (9 – 25)	Rainknow.	Fairly uniform. Fall – spring.
Blue Mountains	25 – 102 (10 – 40)	Snow.	Late fall – early spring; greater than 10% summer convective storms.
Snake River Basin/ High Desert	20 – 64 (8 – 25)	Rainknow.	Fairly uniform with slight peaks in fall and spring.
Northern Rockies	51 – 152 (20 – 60)	Snow.	Fall – spring.

Mountains that also receive most of their precipitation in the late-fall to early spring, as both rain and snow (Swanston 1991). Hydrologically, this transient zone is particularly important during rain-on-snow events. When warm, moist air-masses pass over snowpack, condensation of water on the snow surface occurs, releasing large amounts of latent energy during the phase change of water from vapor to liquid. **A** small amount of condensation can facilitate the rapid melting of substantial volumes of snow, which combined with runoff from rainfall can produce large floods. Snow-dominated systems include those that receive precipitation predominately as snow, including the higher elevations of the Cascade, Sierra Nevada, Olympic, and Rocky Mountain (and associated) ranges, **as** well as mid-elevation interior basins of the Columbia and Snake rivers. In the mountainous regions west of the Cascade and Sierra crests, precipitation is highly seasonal with most falling from fall through spring.

East of the Cascades and Sierra Nevada, the strong seasonal signature of precipitation diminishes and precipitation is spread more evenly throughout the year, particularly where spring and summer convective storms contribute substantially to the total annual precipitation. In high elevation areas of eastern Washington, the Cascades, and the Rocky Mountains, rime and hoar-frost formation may also contribute significantly to the overall water balance of a watershed (Berndt and Fowler 1969; *Gary* 1972; Hindman et al. 1983).

3.6.2 Evapotranspiration

Evapotranspiration losses include those water losses from interception by the canopy and subsequent evaporation, evaporation of water that reaches the soil, and water that enters the soil and is subsequently taken up by plants and transpired back into the atmosphere. The amount of water lost through these processes depends on vegetation type, season, and the nature of the precipitation event, including the intensity, duration, and form of the precipitation, as well as climatological conditions during the event (e.g., temperature, humidity, wind speed).

Interception Losses

Dense coniferous canopies have greater interception storage capacity than those of sparse coniferous forests, deciduous forests, shrublands, or grasslands (Wisler and Brater 1959; Zinke 1967). Rothacher (1963) reported interception and evaporation losses of nearly 100% during low-intensity rainfall events (< 0.13 cm) compared with losses of only 5%–12% during high-intensity events (> 5 cm) in an old-growth Douglas-fir forest in western Oregon. Annual interception losses for woodland-chaparral vegetation in central California ranged from 5%–8%, with seasonal losses of 4% during the winter and 14% during the spring and summer when vegetation was in full foliage (Hamilton and Rowe 1949).

Interception by coniferous canopies during snowfall can also be substantial. Snow may be temporarily stored in the canopy and then delivered to the snowpack during the storm as branches become heavily laden or following the storm by melt or wind action. Satterlund and Haupt (1970) found that 80% of the snow held in the canopy of a forest in Idaho subsequently reached the ground. Only 5% of the total snowfall was lost to interception and subsequent evaporation.

Evaporation Losses

Evaporation directly from the soil or vegetation depends on solar radiation, wind, and vapor pressure gradients between the air and the wetted soil or leaf surface. Vapor pressure gradient in the air is a function of both temperature and humidity. The temperature required for evaporation increases with increasing humidity. Under dense forest canopy, evaporation from wetted soils occurs slowly because of the high degree of shading, low temperatures, relatively high humidity, and low wind speeds typically found in these environments. More open forests allow for greater radiation and higher wind speeds that help remove water vapor from the air-soil interface, maintaining a higher vapor pressure gradient. For soil surfaces exposed to direct solar radiation, evaporation may dry soil more rapidly than transpiration because of high surface temperatures and low humidity (Satterlund and Adams 1992).

Transpiration Losses

Transpiration is the passage of water vapor from living plant tissues into the atmosphere through pores or "stomates." Transpiration rates vary based on a number of plant characteristics, including leaf surface-area, stomatal characteristics, and depth of roots; they also are affected by whether the plants are annual or perennial, and deciduous or coniferous. Coniferous forests generally have the highest leaf surface-area and thereby have the greatest potential for transpiration losses, followed in descending order by deciduous trees, shrubs, grasslands, and desert shrubs. Trees and shrubs with deeper roots can extract moisture from greater depths than grasses and forbes. Coniferous trees in xeric conditions east of the Cascades and Sierra Nevada may have large tap roots that penetrate deep into the soil, allowing moisture to be extracted even during dry periods.

Transpiration rates also depend on climatic conditions including temperature, humidity, and wind speed. In general, transpiration rates increase with increasing temperature; however, stomates will close in response to excessively high or low temperatures, increasing resistance to moisture loss. High humidity reduces the vapor pressure gradient between the plant leaf and the atmosphere, thereby reducing transpiration losses. Winds transport evaporated water vapor away from the leaf surface, thereby maintaining a higher vapor pressure gradient and increasing transpiration.

Soil conditions also influence how much water is available for transpiration. Loam soils tend to have higher water-storage capacity than sandy soils. Similarly, deep soils hold more water than shallow soils. As soil moisture is depleted, the resistance to further uptake by plants increases, and water is supplied to plants at a slower rate (Satterlund and Adams 1992). Insufficient moisture causes closure of leaf stomates, which reduces transpiration losses. Consequently, when soils are moist, transpiration approaches maximum values; when soils are dry, transpiration is substantially less.

Total Evapotranspiration

Estimates of total evapotranspiration losses (interception + evaporation + transpiration) for a number of vegetation communities in the Pacific Northwest indicate that total losses are generally highest for coniferous forest types and slightly lower for chaparral and woodland communities (Table 3-5); however, losses from chaparral, woodland, and semi-arid communities represent a greater percentage of total annual precipitation. This is significant in eastside systems, in part because a substantial amount of precipitation occurs during spring and fall periods when temperatures are warm and evaporation and transpiration rates are high. In contrast, precipitation in the Coast Range and western Cascades generally falls during winter, when transpiration losses are relatively low because of low solar radiation, high humidity, and cool temperatures. These differences between hydrologic processes in eastside versus westside systems are important in determining the potential effects of land-use practices; they are discussed in greater detail in Section **6.1.**

Table 3-5. Estimated precipitation and evapotranspiration for western vegetation communities.

Vegetation community	Precipitation (cm)	Precipitation (inches)	Evapotranspiration (cm)	Evapotranspiration (inches)
Forest				
Lodgepole pine	51–114	20–45	48	19
Engelmann spruce-fir	51–114	20–45	38	15
White pine-larch-fir	64–152	25–60	56	22
Mixed conifer	38–178	15–70	56	22
True fir	51–254	20–100	61	24
Aspen	51–114	20–45	58	23
Pacific Douglas-fir hemlock-redwood	51–254	20–100	76	30
Interior ponderosa pine	51–76	20–30	43	17
Interior Douglas-fir	51–89	20–35	53	21
Chaparral and Woodland				
Southern California chaparral	25–102	10–40	51	20
California woodland-grass	25–102	10–40	46	18
Arizona chaparral	25–51	10–20	43	17
Pinyon-juniper	25–51	10–20	38	15
Semi-arid grass and shrub	13–51	5–20	28	11
Alpine	64–203	25–80	51	20

3.6.3 Infiltration, Subsurface Flow, and Overland Flow

The amount of water that infiltrates into the soil depends on the physical structure of the soil and antecedent moisture conditions. Sandy and gravelly soils derived from colluvium, alluvium, glacial tills, or soils that are rich in organic matter tend to be highly porous and allow rapid infiltration (Swanston 1991). Soils derived from finer-grained materials, including marine and lacustrine materials, or from weathered siltstones, sandstones, and volcanic rocks are less permeable and have lower infiltration capacities. During a given storm event, infiltration capacity decreases through time as soil pores are filled with water (Bedient and Huber 1992). If rainfall intensity (or snowmelt) exceeds infiltration capacities, overland flow occurs. Consequently, the likelihood of overland flow increases with storm intensity and duration.

In forested watersheds, most precipitation reaching the forest floor infiltrates into the soil (Satterlund and Adams 1992). Surface soils in old-growth forests areas typically have high organic content and porosity. Consequently, infiltration capacities are high, and overland flow is uncommon except in areas where soil structure has been modified through human activity or natural disturbance. The majority of water that falls on a forested landscape thereby enters streams via downslope subsurface flow. **As** a result, time of maximum streamflow usually lags behind peak rainfall (Swanston 1991).

In arid and semi-arid systems as well as in deforested lands, vegetation and organic litter are less abundant, and the routing of water once it reaches the soil differs. In areas where the soil surface is exposed, the impact of raindrops can detach and mobilize fine sediments (splash erosion), which settle into soil interstices, creating an impervious surface layer (Wisler and Brater 1959; Heady and Child 1994). As a result of this "rain compaction," a significant proportion of rainfall or snowmelt runs off overland to the stream. Thus, in contrast to forested watersheds, precipitation events in arid and semi-arid systems cause rapid increases in streamflow. This may be particularly evident when soils are further compacted through land-use activities.

3.6.4 Stream Hydrology

Differences in precipitation patterns, evapotranspiration rates, and infiltration processes **lead** to **marked regional differences in hydrologic** regimes of streams. In addition, the size of the drainage basin significantly influences the characteristics of streamflow at a particular point downstream. **As** a general rule, small headwater streams are more hydrologically dynamic than larger streams because runoff occurs more rapidly over steeper areas and because high intensity events are more common in small areas. In the discussion below, we generalize about hydrologic patterns in lower order streams.

Regional Patterns

In the Coast Range, western Cascades, Puget Lowlands, and the Willamette Valley, frequent and heavy precipitation from November to March leads to a highly variable stream hydrograph with multiple peaks that closely correspond to precipitation (Swanston 1991). In the early part of the rainy season, soil moisture is typically low, and a large fraction of rainwater functions to replenish depleted soil moisture. In addition, evapotranspiration rates decrease during the winter as temperatures drop. Consequently, precipitation events of similar intensity will result in higher peak flows in the winter, when soils are more fully saturated and transpiration demands are low, than in the fall. Streamflows are lowest during the summer when precipitation is low, evapotranspiration demands are high, and soil moisture is depleted.

In the transient-snow zone of the mid-elevations of the Cascades and northern Sierra Nevada, soils become saturated **as** rainfall increases in the fall. During the winter, a combination of rain and snow events occur. During rainfall events, water tends to run off quickly to the stream channel because soil moisture is high and evapotranspiration is low. Consequently, increases in streamflow tend to coincide with rainfall. Precipitation that falls as snow is stored above ground for varying lengths of time, but it generally melts within a few weeks of falling (Swanston 1991). Thus, increases in streamflow from melting snow will occur days, or even,weeks after the peak snowfall. Some of the more notable high-flow events occur when substantial snowfall is followed by high-intensity rains. These "rain-on-snow" events can release large volumes of water over short time periods.

In snow-dominated systems—the high Cascades, Sierra Nevada, Blue Mountains and northern Rocky Mountains—moisture from precipitation is stored in snowpack through much of the winter and released when temperatures warm in the late spring. Stream hydrographs are thus characterized by low winter flows followed by rapid increases during the spring snowmelt period. **As** snowpack diminishes, streamflow recedes and summer flows during the dry summer months typically are low, although minor peaks may result from intense convection storms. In the fall, rainstorms of moderate intensity can cause additional peaks in flow (Swanston 1991). Runoff from these events occurs most rapidly in high-elevation areas where soils are shallow and composed of fast-draining colluvial deposits and where transpiration demands are low because of sparse vegetation.

Arid and semi-arid regions east of the Cascades and Sierra Nevada tend to have high numbers of large ephemeral and intermittent stream channels. In part, this is because the timing of precipitation can coincide with periods of relatively high solar radiation in the spring, summer, and fall, unlike west of the Cascades where most precipitation falls during cold, cloudy periods. Much of the precipitation that falls in the warmer months is either rapidly evaporated from the ground or forest canopy or transpired by vegetation. In high intensity events, sudden increases in streamflow can occur where soils are relatively impervious and water is routed rapidly to the stream channel. Those streams that flow year round are generally fed by snowmelt from higher elevations or by ground-water discharge from aquifers recharged during periods of high precipitation.

Floods

Large, infrequent floods play an important role in shaping stream channels through the erosion, transport, and deposition of bed materials. Floods with recurrence intervals of 100 years or more can result in major channel changes, and several decades may be required to re-establish an equilibrium approaching preflood conditions, Some features produced by large floods may last longer than the recurrence interval of the event (Anderson and Culver 1977), implying that large floods may be responsible for specific aspects of valley-floor formation rather than simply acting as disturbance events.

In December 1964, a rain-on-snow storm produced floods with a recurrence interval exceeding 100 years over much of northern California and Oregon. Studies conducted after this event provide information on the geomorphic effects of large floods and on the time needed to achieve a new dynamic equilibrium following such an event. The storm caused numerous debris slides and debris avalanches on slopes, and the resulting flood caused channel erosion and destruction of streamside vegetation (Lisle 1982; Sarna-Wojcicki et al. 1983). Hillslope and valley-bottom erosion put large amounts of sediment into the channel of the Van Duzen River, equal to seventeen times the mean annual sediment input into the channel system (Kelsey 1980). The result **was** a prolonged period of channel aggradation (five to fifteen years), followed by a period of degradation that was not complete in some reaches after twenty years. Erosion and the increased sediment load changed channel morphology, increasing channel width and decreasing channel depth, pool depth, and roughness (Lisle 1982). Floods of magnitude comparable to the 1964 floods occurred throughout the Willamette Valley and in southwestern Washington in February of 1996, resulting in dramatic restructuring of many stream and river channels.

The effectiveness of large floods to shape channel morphology may vary depending on stream size and position in the drainage network as well as on land cover (or recent changes in land cover such as logging). In steep mountain streams, only large, infrequent floods significantly modify valley-floor landforms. In lower gradient alluvial reaches, smaller, more frequent events and ongoing processes modify the valley floor (Grant et al. 1994). Floods also deposit sediments onto the surrounding floodplain, transport and rearrange large woody debris within the channel, clean and scour gravels in streams, recharge floodplain aquifers, and disperse propagules of riparian vegetation.

Droughts

Below-average precipitation and runoff can have significant effects on streams and watersheds. The recent drought in the Pacific Northwest has focused much attention on the health of forest ecosystems east of the Cascade Crest (Quigley 1992). Substantial die-off of forest vegetation has resulted from the synergistic effects of fire suppression and forest practices, which have led to changes in species composition of terrestrial vegetation (see Section 3.3). Drought conditions have weakened trees, making them more vulnerable to infestation by insects or disease. The influence of drought on watershed processes is not well documented; however, it is likely that droughts affect the input of nutrients, allochthonous materials, and large woody debris to stream channels. Within the stream channel, low flows can constrict the available habitat and allow water temperatures to warm, stressing fish or creating thermal barriers that block migration. A potential benefit of drought is that it provides the opportunity for establishment of riparian vegetation within the active stream channel, which in turn can stabilize channel features, dissipate hydraulic energy, and collect sediment when flows rise again (Blau 1995). Tree-ring records from eastside forests indicate that a number of significant droughts lasting from 5 to 20 years have occurred during the past 300 years (Agee 1994).

3.7 Thermal Energy Transfer

Because most aquatic organisms are ectothermic, water temperature plays an important role in regulating biological and ecological processes in aquatic systems. Temperature directly and indirectly affects physiology, development, and behavior of salmonids, as well as mediates competitive interactions, predator-prey relationships, and the incidence of parasitism and disease (see Chapter 4). Land-use practices can significantly change seasonal and diel temperature regimes in streams, primarily through the alteration of forest and riparian canopy but also through irrigation, impoundments, heated industrial effluents, and thermal power plants. In this section, we review the dominant energy transfer processes that are responsible for the heating and cooling of streams, rivers, and lakes to provide the basis for evaluating the effects of land-use practices on salmonid habitat. The role of riparian vegetation in controlling these processes is emphasized.

3.7.1 Heaf Exchange in Streams

Heat energy is transferred to and from streams and rivers by six processes: short-wave radiation (primarily direct solar), long-wave radiation, convective mixing with the air, evaporation, conduction with the stream bed, and advective mixing with inflow from groundwater or tributary streams (Beschta et al. 1987; Sullivan et al. 1990). These processes occur in all streams, but the importance of each process on stream temperatures varies with location and season (Sullivan et al. 1990).

Direct solar radiation is generally the dominant source of energy input to streams and rivers. The amount of solar radiation that reaches and is absorbed by streams and rivers is influenced by season, latitude, topography, orientation of the watershed, local climate, and riparian vegetation. Season and latitude together determine the amount of daylight and the solar angle, both of which affect the amount of energy absorbed by streams (Brown 1980). In mountain or canyon regions, topography may provide substantial shade to streams, particularly at times of the year when the sun is low in the sky and in north-facing drainages. Local climate, and particularly cloud cover, significantly influences how much solar radiation reaches the stream channel. The amount and type of riparian vegetation play dominant roles in regulating incoming solar radiation in smaller streams (Brown 1980; Beschta et al. 1987; Caldwell et al. 1991). The percentage of total solar radiation that reaches the stream surfaces in forested reaches may vary from less than 16% under dense coniferous canopies found in old-growth stands of the Coast Range and western Cascades (Summers 1983) to 28% in old-growth forests east of the Cascades (Anderson et al. 1992, 1993). In alpine, arid, and semi-arid ecosystems, the degree of shading may be less. Deciduous vegetation can provide significant shading during the spring and summer months, but it has minimal effect after leaf drop in the fall. The influence of riparian vegetation on radiation inputs diminishes in a downstream direction. As streams become larger and wider, riparian vegetation shades a progressively smaller proportion of the water surface (Beschta et al. 1987).

Long-wave radiation back into the atmosphere plays a relatively minor role in the overall energy budget of a stream. Long-wave radiation loss is determined primarily by the temperature differential between water and air, with greater exchange occurring when the difference between the air and water temperatures is greatest. Riparian vegetation reduces long-wave radiation through its effect on microclimate within the riparian zone. Temperatures in the riparian zone tend to be cooler during the day and warmer at night than those above the forest canopy; this dampening of diel temperature fluctuations moderates long-wave radiative gains and losses.

Convective and evaporative heat transfer are controlled by temperature and vapor-pressure gradients, respectively, at the air-water interface (Beschta et al. 1987). Greater convective exchange occurs when the temperature differential between air and water is highest. Similarly, evaporative losses are highest at low humidity. Wind facilitates both convective and evaporative losses by displacing air near the air-water interface as it approaches thermal equilibrium with the water and as it becomes more saturated through evaporation. Riparian vegetation modifies convective and evaporative heat-exchange losses by creating a microclimate of relatively high humidity, moderate temperatures, and low wind speed compared with surrounding uplands. These microclimate conditions tend to reduce both convective and evaporative energy exchange by minimizing temperature and vapor-pressure gradients.

Conductive transfer of heat generally represents a minor component of a stream heat budget. The amount of heat transferred depends on the nature of the substrate, with bedrock substrates being more efficient in conducting heat than gravel beds (Beschta et al. 1987). Brown (1980) estimates that heat flow into bedrock stream beds may be as high as 15%–20%of the incident heat. Heat that is transferred to the streambed during the daylight hours serves to heat streams during periods of darkness, thereby dampening diel fluctuations. In shallow, clear streams, without shade from riparian vegetation, solar energy may penetrate through the water column and heat the substrate directly.

The role of advection depends on the volume of groundwater or tributary inputs relative to the total stream discharge; consequently, the importance of advection tends to diminish in a downstream direction. Nevertheless, even when groundwater inputs to streams are small, they may provide thermal heterogeneity that is biologically important (see Sections 5.2.1 and 5.2.3). In addition, certain regions east of the Cascade Range (e.g., the Deschutes Basin) are underlain with porous basaltic formations that absorb large amounts of water during periods of high runoff and release it later in the year. These groundwater inputs can significantly moderate streamflow and temperature regimes in both summer and winter.

As subsurface flow moves laterally and downward towards stream beds, water temperatures equilibrate with those in the subsurface soil layers (Beschta et al. 1987); consequently, the temperature of water that enters streams from groundwater flow depends on ambient conditions in the soil environment. Surface-soil temperatures follow seasonal air temperature patterns with a time lag that increases with increasing depth (Meisner 1990). Seasonal fluctuations are greatest at the surface and decrease with depth down to the "neutral zone," generally about 16–18 m below the surface, where temperatures remain constant throughout the year (Meisner 1990). If the groundwater flow originates below the neutral zone, then groundwater temperatures will remain constant; if it originates above the neutral zone, then groundwater temperatures will exhibit seasonal variation (Meisner 1990). Melting snow infiltrates into the soil at temperatures approaching 0°C in snow-dominated systems (Beschta et al. 1987).

3.7.2 Stream Temperature Regulation

All of the above processes interact to produce the temperature regimes observed in streams and rivers; however, the relative importance of each process differs among locations. In small- to intermediate-sized streams of forested regions, incoming solar radiation represents the dominant form of energy input to streams during summer, with convection, conduction, evaporation, and advection playing relatively minor roles (Brown 1980; Beschta et al. 1987; Sullivan et al. 1990). Groundwater inputs may be important in small streams where they constitute a large percentage of the overall discharge, particularly during periods of the year when flows are low. Downstream, where flow increases, the effects of riparian shading and advective mixing generally diminish, and the importance of evaporative heat-loss increases.

Channel characteristics may also significantly affect heat-exchange processes. The amount of heat that is gained or lost and the rate at which exchange takes place depend on the surface area of the stream or river. Wide, shallow streams exhibit greater radiative, convective, and evaporative exchange and, consequently, heat and cool more rapidly than deep, narrow streams. Similarly, the rate of energy exchange is affected by seasonal changes in stream discharge, which alter surface-to-volume ratios and determine the relative importance of groundwater inputs. In most streams in the Pacific Northwest, groundwater inputs are critical to cool streams during warm summer months. Regional differences in stream temperatures result from differences in climatic factors (e.g., humidity, air temperature). Streams in the Coast Range and western Cascades are moderated by the maritime climate and undergo smaller seasonal temperature fluctuations than those in the continental climates east of the Cascades. Elevation also influences stream temperatures, primarily because of elevational gradients in air temperatures that lead to greater convectional heating (Beschta et al. 1995). Finally, high turbidity in streams and rivers substantially increases the absorption of high-energy, shortwave radiation (Wetzel 1983) and thereby can affect stream heating.

3.7.3 Lakes and Reservoirs

Lakes and reservoirs are heated primarily by incoming solar radiation, although some heat is transferred by convection, conduction (in shallow waters), and evaporation (Wetzel 1983). In clear water, over one-half of the incoming solar radiation is absorbed in the upper two meters of water, and more may be absorbed in waters with high turbidity. In temperate lakes, incoming solar radiation exceeds outgoing long-wave radiation during the summertime, and water at the surface is gradually warmed. Because warm water is less dense than cold water, it tends to remain near the surface and is resistant to mixing by the wind. As a result, thermal stratification can occur with a warm and relatively well-mixed "epilimnion" overlying a cooler "hypolimnion." Between these two layers is a transition zone, or "metalimnion," where temperatures rapidly decrease with increasing depth. During the fall as solar radiation decreases, temperatures in the surface layers cool, and the mixing of epilimnetic and hypolimnetic waters occurs as they reach comparable temperature and density. During the winter, lake waters tend to remain mixed except where temperatures are sufficiently cold to cause freezing. Because water reaches maximum density at 4°C, lakes that freeze are colder near the surface and warmer near the bottom (inverse stratification). When lakes become ice-free in the spring, density is relatively uniform, and mixing of the water column will occur again provided there is sufficient wind at the surface.

The above pattern is characteristic of deeper lakes in the Pacific Northwest. In shallower lakes and ponds, lakes may turn over many times each year, whenever high wind conditions occur. Such systems are usually poor habitat for salmonids because they warm throughout the water column.

Thermal structure plays an important role in determining the distribution and production of aquatic organisms within a lake or reservoir. Stratification of lakes may restrict the habitats of fishes and other aquatic organisms. Moreover, during the spring and fall mixing periods, the circulation (turnover) of water brings nutrient-rich waters to the surface and stimulates production of phytoplankton and zooplankton. Release of either epilimnetic or hypolimnetic water from stratified reservoirs can markedly influence downstream temperature regimes in ways that may have adverse consequences for fish.

3.8 Nutrient Cycling/Solute Transport

Water is the major agent for the flux of dissolved and particulate matter across the landscape, integrating processes of chemical delivery in precipitation, geologic weathering, erosion, chemical exchange, physical adsorption and absorption, transport and retention in surface waters, and biotic uptake and release. At any point within a landscape or catchment, concentrations of nutrients or suspended material result from many abiotic and biotic processes.

Geology, climate, and biological processes across a landscape determine patterns of nutrient cycling. The primary determinant of the chemistry of most surface waters is the composition and age of the parent geology. The major rock types—igneous, sedimentary, and metamorphic—have characteristic compositions of major cations and anions, as well as minor chemical constituents that serve as nutrients (e.g., nitrogen, phosphorus) for biota. The high temperatures and pressure under which igneous and metamorphic rocks are formed alters the chemical composition by volatilizing elements and compounds that are released as gases (e.g., nitrogen, inorganic carbon) to the atmosphere. Sedimentary rocks contain minerals that have been weathered from other sources previously, and they may contain greater amounts of biologically derived material because of the less harsh conditions of their formation. Geochemistry of the parent material governs rates of dissolution or weathering and, thus, influences concentrations of dissolved chemicals in surface waters.

Climate strongly influences general surface-water chemistry and nutrient concentrations through two major processes—direct input of chemicals through precipitation and influence on hydrology (Gibbs 1970). The atmosphere is a major source of elements and compounds. Weather patterns affect the available source areas for water and chemicals in the atmosphere and subsequent precipitation that falls on land. Both natural and anthropogenic sources may create distinctive chemical signatures in precipitation. Climate also determines the general hydrologic regime and establishes physical conditions that influence evaporative losses of water. Arid areas typically exhibit high concentrations of dissolved ions because of high rates of evaporation and subsequent concentration of chemicals in solution. The hydrologic regime is a function of climate and geographic features of the landscape, and it is a major determinant of weathering rates, dilution, and timing of nutrient transport. Patterns in runoff may be mirrored by differences in surface water chemistry. The flashy flow-regimes of rain- and rain-on-snow dominated systems create a similar episodic pattern in nutrient transport, while the more steady flow regimes of snow-dominated systems produce more predictable nutrient transport patterns.

The biota of terrestrial, riparian, and aquatic ecosystems strongly influence the cycling of major nutrients and associated chemical parameters (Likens et al. 1977; Meyer et al. 1988) through such processes as photosynthesis, respiration, food consumption, migration, litter fall, and physical retention. Surface waters are exposed to various sources of inputs, sites of biological uptake, and surfaces for physical exchange (Gregory et al. 1991). Stream substrates serve as sites for colonization and attachment by aquatic organisms ranging from microbes to vertebrates. Many aquatic organisms have distinct substrate relationships; therefore, the composition of the stream bed can directly influence nutrient cycling. Organic substrates, such as leaves and wood, create important sources for microbial colonization and subsequent nutrient cycling (Aumen et al. 1985a, 1985b; Meyer et al. 1988). These organic substrates also serve as sources of dissolved organic carbon for microbial activity or transport into the water column (Dahm 1981). Woody debris in particular plays a critical role as a food resource, substrate, site of physical exchange, site for biological uptake, and roughness element that reduces water velocity and increases retention (Harmon et al. 1986). Land-use practices typically alter the organic substrates of stream channels, and thereby influence water quality.

Vegetated floodplains along streams and rivers as well as mudflats and vegetation beds in estuaries create a mosaic of geomorphic surfaces and riparian plant communities (Fonda 1974; Gregory et al. 1991; Bayley and Li 1992). Floodplains influence the delivery and transport of material by 1) delivering stored material during high flows, 2) retaining material in transport from the main channel, 3) providing a matrix of sediment for subsurface flow, and 4) reducing velocities of water and increasing the potential for retention. Elimination of floodplains greatly reduces the assimilative and storage capacity of a stream system **and** is one of the major forms of anthropogenic alteration of nutrient cycling in lotic ecosystems (Smith et al. 1987; Junk et al. 1989; Sparks et al. 1990). Side channels on floodplains and in estuaries are habitats with extensive contact with the water column and lower velocities than the main channel; consequently, these lateral habitats typically exhibit high rates of nutrient uptake and biological productivity (Cooper 1990).

Streamside forests, estuarine vegetation beds (tidal marshes), and other plant communities create a filter through which nutrients in solution must pass before entering surface waters (Pionke et al. 1988; Gregory et al. 1991). Retention of nutrients in groundwater is a critical component of nutrient cycling within a basin (Simmons et al. 1992). Commonly, these vegetative corridors remove 60%–90% of the nitrogen and phosphorus in transport (Lowrance et al. 1984; Peterjohn and Correll 1984; Lowrance 1992). Modification of riparian forest structure can substantially change long-term patterns of nutrient cycling within a catchment (Pinay et al. 1992).

One of the most overlooked components of a stream and its valley is the hyporheic zone, the area of flow beneath the surface of the stream bed (Stanford and Ward 1988; Bencala 1993). In alluvial valleys, the hyporheic zones may extend several meters below the channel bed, as well as a kilometer or more laterally. Recent research indicates the hyporheic zone plays important roles in nutrient cycling, temperature modification, dissolved oxygen microbial processes, meiofaunal communities and refugia for a wide range of organisms (Pinay and Decamps 1988; Stanford and Ward 1988; Triska et al. 1990; Valett et al. 1990; Hendricks and White 1991). In many streams, **as** much as 30%–60% of the flow occurs in the hyporheic zone and may exceed these levels in porous bed materials or during low flow conditions. The majority of nutrient uptake in streams may occur in the hyporheic zone in desert, forest, or grassland ecoregions (Duff and Triska 1990).

3.8.1 Major Chemical Species and Dissolved Nutrients

Surface waters contain a complex array of major chemical species, biologically important nutrients, and numerous trace elements and compounds. The major dissolved constituents include cations and anions that are required by living organisms but are so abundant that they rarely limit biological production. In addition, surface waters contain concentrations that they limit rates of production of plants, microbes, or consumers. The major nutrients or macronutrients are nitrogen, phosphorus, and carbon. Micronutrients are generally required in such low amounts that their availability is rarely limiting, but studies over the last several decades have demonstrated that the productivity of some systems may be limited by micronutrients and many processes are commonly limited by the availability of these chemicals. This review only covers the macronutrients.

The major cations in surface waters include the divalent cations of calcium and magnesium and the monovalent cations of sodium and potassium. In general, the order of dominance in surface waters of the world is $\mathrm{Ca^{++}} > \mathrm{Mg^{++}} > \mathrm{Na+} > \mathrm{K^{+}}$, but local geology can alter their relative abundance (Gibbs 1970). These elements play critical roles in all biological systems as well as influence the reactivity and abundance of other elements. The exchange of these cations, either physically or through biological absorption, can alter the availability of hydrogen ions and thereby alter pH, which strongly influences biota and fundamental ecological processes.

The major anions in surface waters consist of the divalent anions of carbonate and sulfate and the monovalent anions of bicarbonate and chloride (Gibbs 1970). In temperate waters, the dominance of anions is ordered: $\mathrm{HCO_3^{-}} > \mathrm{CO_3^{--}} > \mathrm{SO_4^{--}} > \mathrm{Cl^{-}}$. Inorganic carbon and sulfate are biologically important in all ecosystems, and the inorganic carbon species largely determine the buffering capacity and, consequently, the pH conditions of surface waters.

Nitrogen

Nitrogen exists in solution as both inorganic forms—nitrogen gas (N,), nitrate ($\mathrm{NO_3^{-}}$), nitrite ($\mathrm{NO_2^{-}}$), ammonia ($\mathrm{NH_3}$), or ammonium ($\mathrm{NH_4^{+}}$)—and organic forms (organic N). In many areas of the Pacific Northwest, surface waters commonly have extremely low concentrations of dissolved nitrogen because of the underlying volcanic parent geology, which was created under intense temperature and pressure (Thut and Haydu 1971; Sollins and McCorison 1981; Norris et al. 1991).

Biological processes largely mediate the different forms of nitrogen (Gosz 1981). Nitrogen fixation converts $\mathrm{N_2}$ into $\mathrm{NH_3}$ under anaerobic conditions or in specialized cells, and organisms subsequently use the ammonia to form amino acids and proteins. Organic nitrogen is metabolized to ammonium as a waste product or microbial decomposition converts organic N to ammonium through the process of ammonification. Certain microorganisms are capable of oxidizing ammonia to nitrite or nitrate. Plants and heterotrophic microorganisms can then reduce nitrate to form ammonia and subsequently proteins and amino acids. Under anoxic conditions, certain microorganisms can reduce $\mathrm{NO_3^{-}}$ to N,. These transformations create intricately linked cycles of nitrogen, and under nitrogen-limited conditions, these links are tightly coupled. **As a** result, certain forms—such as ammonia or nitrate—are rarely present naturally in high concentrations because they are so rapidly incorporated into other nitrogenous molecules or are modified.

Riparian areas play major roles in nitrogen cycling by providing year-round anaerobic conditions (Green and Kauffman 1989; Mulholland 1992). Rates of denitrification (and nitrogen fixation) are enhanced in the anaerobic conditions and in the high moisture and organic substrates that denitrifying bacteria require (Myrold and Tiedje 1985; Ambus and Lowrance 1991; Groffman et al. 1991). Rates of denitrification in riparian soils in the Cascade Mountains of Oregon are four to six times higher than in upslope forests, and alder-dominated reaches exhibit the highest observed rates (Gregory et al. 1991). Alder is a common streamside plant and is also a nitrogen fixer; thus, alder-dominated riparian areas are potential sources of nitrogen in stream water (Tarrant and Trappe 1971). As noted above, elevated rates of denitrification may negate the contribution of alders, but it is possible for extremely high concentrations of nitrate (> 5 mg NO_3-N/l) to occur where litter inputs are high and water velocities are low (Taylor and Adamus 1986). These conditions have been observed primarily in the Coast Range where alder may extend from stream's edge to the ridgeline.

Catchments generally process nitrogen efficiently because it is such a biologically important element. A small western basin retained approximately 99% of the nitrate that entered in precipitation (Rhodes et al. 1985). Loss of nitrogen from terrestrial ecosystems is mediated by uptake in the aquatic ecosystems, particularly in nitrogen-limited ecosystems, such as the basalt-dominated Pacific Northwest (Triska et al. 1982, 1984). Studies of nitrogen uptake in streams of the Cascade Mountains indicate that approximately 90% of the nitrate or ammonium introduced into stream water is assimilated within 500–2,000 m, depending on the size of the stream (Lamberti and Gregory 1989; D'Angelo et al. 1993).

Phosphorus

Phosphorus in surface waters is largely derived from mineral sources. Inorganic phosphorus includes many compounds incorporating the phosphate ion ($PO_4^{- - -}$). Concentrations of inorganic phosphorus are low in many geologic areas, and as a result, phosphorus commonly is a limiting nutrient for primary production and microbial processes (Wetzel 1983). In much of the Pacific Northwest, however, the basaltic parent geologic material contains abundant and relatively easily weathered forms of inorganic phosphorus; thus, concentrations of phosphorus in streams and rivers commonly exceed 10 μg $PO_4\text{–}P \cdot L^{-1}$ (Fredriksen et al. 1975; Salminen and Beschta 1991; Bakke 1993).

3.8.2 Nutrient Spiraling and Retention

Nutrient cycling is often viewed as a closed system in which chemicals pass through various states and reservoirs within the ecosystem of interest. Stream ecosystems present an interesting contrast to this perspective because of their unidirectional flow from headwaters to large rivers to the ocean. The Nutrient Spiraling Concept was developed to more accurately represent the spatially dependent cycling of nutrients and the processing of organic matter in lotic ecosystems (Newbold et al. 1982; Elwood et al. 1983).

The longitudinal nature of streams and rivers strongly influences patterns of nutrient uptake. In the Nutrient Spiraling Concept, one complete cycle of a nutrient depends upon the average distance a nutrient atom moves in the water compartment (i.e., the uptake length), the average distance a nutrient atom moves in the particulate compartment, and the average distance a nutrient atom moves in the consumer compartment. The Nutrient Spiraling Concept provides a useful framework to investigate the dynamics of dissolved and particulate material in streams and rivers (Mulholland 1992). Alteration of riparian areas, stream channels, and biotic assemblages can be viewed in terms of changes in flux and uptake, the two major components of spiraling length. Efficiency of nutrient use can be quantified in terms that are relevant to the cycling of nutrients along a river valley or drainage network.

Downstream transport of dissolved or particulate material is a complex function of physical trapping, chemical exchange, and biological uptake (Minshall et al. 1983; Speaker et al. 1984). Retention of material in streams is not necessarily uniform along a reach of stream. Physical discontinuities, such as debris dams, boulders, pools, and sloughs, alter retention patterns. The ionic strength or salinity of surface water tends to increase from headwaters to large rivers, reflecting the accumulation of weathering products and material produced by terrestrial and aquatic ecosystems (Dahm et al. 1981). Spiraling length increases and retention efficiency decreases as streams become larger because of the decreased friction, increased average velocity, and lower probability of being trapped by bed material. This pattern is moderated in braided channels and at high flows as streams flow out of their banks and are slowed by the roughness of adjacent forests and floodplains (Welcome 1988; Junk et al. 1989; Sparks et al. 1990).

Different environments may alter retention patterns for dissolved and particulate matter. Areas of intense biological activity increase biotic uptake and alter patterns of retention. Simplification of stream ecosystems will tend to make longitudinal patterns of retention more uniform and less efficient, thus lowering biological productivity.

Retention of material represents a fundamental ecological feature that integrates the supply and use of nutrients and food resources. Historically, salmon and lamprey carcasses brought significant quantities of organic matter from the Pacific Ocean into freshwater ecosystems of the Pacific Northwest (Bilby and Bisson 1992). The abundance of salmon carcasses has long been correlated with the productivity of sockeye lakes in Alaska for the subsequent year class (Donaldson 1967). Recent studies have demonstrated that as much as 30% of the nitrogen for higher trophic levels in streams in the Pacific Northwest may be derived from marine ecosystems (Bilby et al. 1996). In addition, retention of carcasses in streams has been linked to channel complexity and abundance of woody debris (Cederholm and Peterson 1985). Declines in anadromous fishes in the Pacific Northwest may signal more fundamental changes in productivity of stream ecosystems than the simple loss of stocks or species.

Disturbances can accelerate or slow the loss of nutrients and the efficiency with which terrestrial and aquatic ecosystems use them (Vitousek and Melillo 1979; Beschta 1990). Generally, such disturbances disrupt nutrient cycling over the short-term (i.e., less than a decade) (Resh et al. 1988), but as ecosystems recover, they more efficiently cycle available nutrients. Many disturbances also increase habitat complexity (Swanson et al. 1982a), thereby increasing the efficiency of retention after an initial recovery period (Bilby 1981; Aumen et al. 1990). The frequent disturbances associated with stream ecosystems make them one of the most dynamic ecosystems with respect to nutrient cycling and biotic community organization (Minshall et al. 1985; Minshall 1988). Changes in community organization and process rates in response to changes in long-term nutrient availability may not be fully exhibited for years (Stottlemyer 1987; Power et al. 1988; Peterson et al. 1994).

3.9 Roles of Riparian Vegetation

Riparian zones constitute the interface between terrestrial and aquatic ecosystems (Swanson et al. 1982b; Gregory et al. 1991), performing a number of vital functions that affect the quality of salmonid habitats as well as providing habitat for a variety of terrestrial plants and animals. While processes occurring throughout a watershed can influence aquatic habitats, the most direct linkage between terrestrial and aquatic ecosystems occurs in the riparian area adjacent to the stream channel. Consequently, the health of aquatic systems is inextricably tied to the integrity of the riparian zone (Gregory et al. 1991; Naiman et al. 1992).

Riparian vegetation provides numerous functions including shading, stabilizing streambanks, controlling sediments, contributing large woody debris and organic litter, and regulating the flux and composition of nutrients (FEMAT 1993; O'Laughlin and Belt 1994; Cederholm 1994). Riparian-aquatic interactions are now recognized by scientists as so important that riparian buffers have been established as a central element of forest practices rules and watershed restoration efforts. Authors in several recent publications have advocated a functional approach to riparian management, attempting to identify "zones of influence" for critical riparian processes (McDade et al. 1990; FEMAT 1993; O'Laughlin and Belt 1994). These approaches recognize that the influence of riparian vegetation on stream ecosystems generally diminishes with increasing distance from the stream channel. In this section, we review principal functions of riparian vegetation and summarize the available literature on zones of riparian influence. Riparian zones of influence and effective riparian buffer widths are elaborated further in Part II, Section 14.2.3.

3.9.1 Shade

In small headwater streams, riparian vegetation moderates the amount of solar radiation that reaches the stream channel, thereby dampening seasonal and diel fluctuations in stream temperature (Beschta et al. 1987) and controlling primary productivity. The effectiveness of riparian vegetation in providing shade to the stream channel depends on local topography, channel orientation and width, forest composition, and stand age and density (Beschta et al. 1987; FEMAT 1993). Naiman et al. (1992) report that in westside forests the amount of solar radiation reaching the stream channel is approximately 1%–3% of the total incoming radiation for small streams and 10%–25% for mid-order streams. In winter, streamside vegetation provides insulation from radiative and convective heat losses (see Section 3.7.1), which helps reduce the frequency of anchor-ice formation (Murphy and Meehan 1991). Thus, riparian vegetation tends to moderate stream temperatures year round. The numerous biological and ecological consequences of elevated stream temperatures on salmonids include effects on physiology, growth and development, life history patterns, competitive and predator-prey interactions, and disease (see Section 4.3).

The FEMAT (1993) report presents a generalized curve relating cumulative effectiveness of the riparian canopy in providing shade relative to distance from the stream channel for westside forests (Figure 3-2).

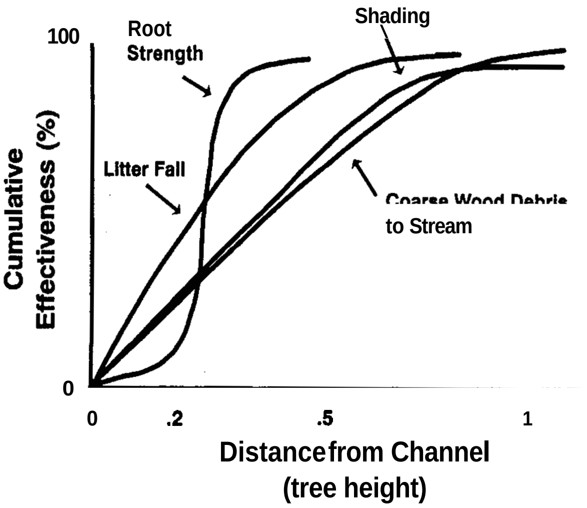


Figure 3-2. Riparian forest effect on streams as a function of buffer width. From FEMAT (1993).

They propose that for these systems close to 100% of the potential shade value (rarely complete canopy cover) can be maintained by buffer widths equal to one site-potential tree height (i.e., the potential height of a mature tree at the particular location). In the Oregon Coast Range and western Cascades, buffer zones of 100 feet or more can provide as much shade as intact old-growth forests (Brazier and Brown 1973; Steinblums et al. 1984). Similar assessments for eastside forests as well as arid and semi-arid shrublands have not been published; effective buffers widths in these systems may differ substantially.

3.9.2 Bank Stabilization

Riparian vegetation increases streambank stability and resistance to erosion via two mechanisms. First, roots from woody and herbaceous vegetation bind soil particles together, helping to maintain bank integrity during erosive high-streamflow events (Swanson et al. 1982b). Diverse assemblages of woody and herbaceous plants may be more effective in maintaining bank stability than assemblages dominated by a single species; woody roots provide strength and a coarse root network, while fine roots fill in to bind smaller particles (Elmore 1992). The root matrix promotes the formation of undercut banks, an important habitat characteristic for many salmonids (Murphy and Meehan 1991). Second, stems and branches moderate current velocity by increasing hydraulic roughness. East of the Cascades, grasses, sedges, and rushes tend to lie down during high flows, dissipating energy and protecting banks from erosion (Elmore 1992).

Riparian vegetation may also facilitate bank-building during high flow events by slowing stream velocities, which in turn helps to filter sediments and debris from suspension. This combing action helps to stabilize and rebuild streambanks, allowing the existing channel to narrow and deepen, and increasing the effectiveness of riparian vegetation in providing bank stability and shade (Elmore 1992). During overbank flows, water is slowed and fine silts are deposited in the floodplain, increasing future productivity of the riparian zone.

Vegetation immediately adjacent to the stream channel is most important in maintaining bank stability. The FEMAT (1993) report suggests that the role of roots in maintaining streambank stability is negligible at distances of greater than 0.5 tree heights from the stream channel (Figure 3-2). In wide valleys where stream channels are braided, meandering, or highly mobile, the zone of influence of root structure is substantially greater.

3.9.3 Sediment Control

The regulation of sediment flow is a major function of the riparian zone. Riparian vegetation and downed wood in the riparian zone can reduce the amount of sediment delivered from upland areas to the stream channel in several ways. By providing physical barriers, standing or downed vegetation can trap sediments moving overland during rainfall events. Riparian zones, however, are less effective in regulating channelized erosion. Most surface erosion occurs in channelized flows that may travel thousands of feet (Belt et al. 1992; O'Laughlin and Belt 1994). Thus, riparian vegetation may have little influence on sediments derived from outside of the riparian zone. Riparian vegetation may also influence sediment inputs by reducing the likelihood of mass failures along the stream channel through the stabilizing action of roots and by buffering the stream from mass wasting that initiates in upland areas, although riparian vegetation may have little effect during large, deep-seated landslides (Swanson et al. 1982b).

The zone of riparian influence for sediment regulation is difficult to define because of different ways sediment may enter the stream channel. The FEMAT (1993) review of the literature suggests that riparian zones greater than 200 feet (i.e., about one site-potential tree height) from the edge of the floodplain are probably adequate to remove most sediment from overland flow. However, O'Laughlin and Belt (1994) suggest sediment control cannot be achieved through riparian zones alone because channel erosion and mass wasting are significant sources of sedimentation in forested streams. For these events the zone of influence may extend several hundred meters from the floodplain (FEMAT 1993), depending on the soil type, slope steepness, and other factors that influence the susceptibility of hillslopes to mass wasting or channelized erosion.

3.9.4 Organic Litter

Riparian vegetation contributes significant quantities of organic litter to low- and mid-order streams. This litter constitutes an important food resource for aquatic communities (Naiman et al. 1992). The quality, quantity, and timing of litter delivered to the stream channel depends on the vegetation type (i.e., coniferous versus deciduous), stream orientation, side slope angle, stream width, and the amount of stream meander (Cummins et al. 1994). In conifer-dominated riparian zones, 40%–50%of the organic litter consists of low quality cones and wood, which may take several years to decades to be processed. In contrast, high quality material from deciduous forests may decay within a year. Although conifers have the greater standing biomass, shrub- and herb-dominated riparian assemblages provide significant input in many streams (Gregory et al. 1991). Over 80% of the deciduous inputs, primarily leaves, are delivered during a 6–8 week period in the fall (Naiman 1992), while coniferous inputs are delivered throughout the year (Cummins et al. 1994).

The extent of the riparian zone of influence for organic-litter inputs depends on geomorphology and stream size. Upland forests beyond the riparian zone can contribute litter to small streams in steep basins through direct leaf-fall and overland transport of material by water. Larger streams (3rd to 5th order) are more influenced by vegetation in the immediate riparian zone. Large lowland streams tend to have complex floodplain channels with minimal upland interactions. However, the lateral movement of unconstrained alluvial channels effectively increases the potential riparian zone of influence. In westslope forested systems, most organic material that reaches the stream originates within 0.5 tree heights from the stream channel (Figure 3-2) (FEMAT 1993). Vegetation type may also influence the riparian zone of influence because deciduous leaves may be carried greater distances by the wind than coniferous litter.

3.9.5 Large Woody Debris

Large woody debris (LWD) provides critical structure to stream channels, although full recognition of the importance of large wood in stream ecosystems has only come in the last 20 years (Swanson et al. 1976; Swanson and Lienkaemper 1978; Harmon et al. 1986). For more than 100 years, large wood was removed from stream channels in the United States to facilitate boat traffic and the floating of **logs** downstream. In addition, up until the late 1970s and early 1980s, biologists viewed large wood as an impediment to fish migration and recommended clearing woody debris from stream channels (Sedell and Luchessa 1981). Consequently, the many roles of large wood in streams, from small headwaters to large river systems, have been greatly diminished over time.

Large wood enters the stream channel through two different pathways: the steady toppling of trees as they die or are undercut by streamflow, and catastrophic inputs associated with windstorms, mass failures, and debris torrents (Bisson et al. 1987; Cummins et al. 1994). Once in the stream channel, large woody debris influences coarse sediment storage; increases habitat diversity and complexity, gravel retention for spawning habitat, and flow heterogeneity; provides long term nutrient storage and substrate for aquatic invertebrates; moderates flow disturbances; increases retention of allochthonous inputs, water, and nutrients; and provides refugia for aquatic organisms during high- and low-flow events (Bisson et al. 1987). The ability of large wood to perform these functions depends in part on the size and type of wood. In general, the larger the size of the debris, the greater its stability in the stream channel, since higher flows are needed to displace larger pieces (Bilby and Ward 1989). In addition, coniferous logs are more resistant to decay than deciduous logs and hence exhibit greater longevity in the stream channel (Cummins et al. 1994).

Although LWD performs essential functions in all streams, the relative importance of each of the processes listed above varies with stream size. In small, steep headwater streams (1st and 2nd order), large volumes of stable LWD tend to dominate hydraulic processes. Generally, woody debris is large enough to span the entire channel, resulting in a stepped longitudinal profile that facilitates the formation of plunge pools downstream of obstructions (Grant et al. 1990). This stepped profile increases the frequency and volume of pools, decreases the effective streambed gradient, and increases the retention of organic material and nutrients within the system, thus facilitating biological processing (Bisson et al. 1987). Woody debris within the channel increases velocity heterogeneity and habitat complexity by physically obstructing the streamflow, creating small pools and short riffles (Swanston 1991). Diverted currents create pools (plunge, lateral, backwater) and riffles, flush sediments, and scour streambanks to create undercut banks (Cummins et al. 1994). In sediment-poor systems, LWD retains gravels that are essential for spawning salmonids. Larger debris-dams store fine sediment and organic materials, reducing their rate of transport downstream. In addition debris-dams protect the downstream reaches from rapid changes in sediment loading, which may degrade spawning gravels, fill pools, and reduce invertebrate populations.

In mid-order streams, large woody debris functions primarily to increase channel complexity and flow heterogeneity by 1) anchoring the position of pools along the thalweg, 2) creating backwaters along the stream margin, **3)** causing lateral migration of the channel, and 4) increasing depth variability (Maser et al. 1988). Large wood deposits tend to occur along margins, or in mid-channel where physical obstructions such as gravel bars collect wood (Bisson et al. 1987). Bilby and Ward (1989) examined streams in western Washington and found a number of differences in the roles of large wood in relation to stream size. Average diameter, length, and volume of pieces of wood were generally greater in mid-order streams than in low-order streams. Large wood was important in pool formation in mid-sized streams; however, these were more likely to be debris-scour pools than plunge pools. In addition, the ability of wood to accumulate sediment diminished as streams became larger, a result of increased stream power. Distributions of organisms associated with woody debris, including various salmonids, changed relative to the changes in woody debris distribution along the stream channel. Other important functions of large wood in mid-order streams include the retention of salmon carcasses and organic detritus, which provide nutrients to the flora and fauna within the stream and in the adjacent riparian area (Bilby et al. 1996).

The role of large woody debris in high order streams is generally less well documented; however, historical records indicate that large debris jams once played a major role in floodplain and channel development on major rivers, such as the Willamette River in Oregon and other systems in the Puget Lowlands of Washington (Sedell and Luchessa 1981). In these high-order streams, large woody debris increased channel complexity by creating side channels, backwaters, and ponds, as well as refugia for aquatic organisms during winter storm events. During high flows, sediments were deposited on the floodplains and in riparian zones, increasing the productivity of these soils. Extensive snag removal and channelization over the last 100 years have diminished these roles of wood in larger river systems. Today, solitary pieces of woody debris are generally not large enough to span the active channel or substantially modify flows (Maser et al. 1988), although woody debris along the outside bends of river banks provides habitat in an otherwise simplified habitat zone (Swanston 1991). Wood snags that remain in the main channel are utilized by insects and fish, particularly in larger river systems with unstable sand substrate (Marzolf 1978; Benke et al. 1984).

Defining the zone of influence for input of all sources of large woody debris is difficult because methods of delivery differ. Most wood likely enters the stream from toppling or windthrown trees; however, wood may also enter the channel through mass wasting and debris torrents. The likelihood that a falling tree will enter the stream channel depends on tree height, distance from the stream channel, and the nature of the terrain. On level terrain, the direction that a tree will fall is essentially random (Van Sickle and Gregory 1990; Robison and Beschta 1990b) except along streambanks, where undercutting causes trees to lean and fall in the direction of the channel. On steep terrain, however, there is generally a higher probability that the tree will fall downslope into the stream channel. The greatest contribution of large wood to streams comes from trees within one tree height of the channel that topple into the stream (Figure 3-2) (FEMAT 1993). McDade et al. (1990) found that source distances of LWD were as far as 55 m from the stream channel in old-growth forests of the Coast and Cascade ranges (OR and WA) with average tree heights of 57.6 m. Murphy and Koski (1989) found that most (99%) large wood (pieces > 3 m in length) in southeastern Alaskan streams originated within 30 m of the channel (approximately 0.75 tree heights). For episodic inputs of large woody debris via mass wasting and debris torrents, defining the zone of influence becomes more difficult. The likelihood of wood entering the stream will vary with conditions that control the frequency of mass wasting, including the slope, soil type, and hydrology. Assessing appropriate zones of influence for these events is probably beyond our current level of scientific understanding. Cummins et al. (1994) and Reeves et al. (1995) report that O-order channels generate most landslides containing trees and coarse sediments.

3.9.6 Nutrients

Riparian zones mediate the flow of nutrients to the stream and are, therefore, important regulators of stream production. Subsurface flow from upland areas carries nutrients and dissolved organic matter to the riparian zone, where these materials are taken up by vegetation for plant growth or are chemically altered (Naiman et al. 1992). Lowrance et al. (1984) found that even narrow riparian zones along streams in agricultural lands significantly affected stream chemistry. Riparian forests modify the chemical composition and availability of carbon and phosphorus, and they promote soil denitrification through changes in the position of oxic-anoxic zones (Pinay et al. 1990 in Naiman et al. 1992). During overbank flows, nutrients from floodwaters may be absorbed by riparian vegetation, reducing the total nutrient load in the stream (Cummins et al. 1994). Dissolved organic matter inputs can occur from numerous sources besides groundwater. These include leachate from entrained litter and large woody debris in the channel, algal, invertebrate, and fish excretions; and floodplain capture at the time of inundation (Gregory et al. 1991).

We found no published attempts to define zones of influence for nutrient cycling. Most likely, this reflects the difficulty in tracing the movement of nutrients, particularly with those elements such as nitrogen for which the number of alternative pathways is great. As discussed in Section 3.8, conditions throughout the watershed influence stream chemistry; consequently, the zone of influence extends to the top of the watershed, even though it may be years before nutrients ultimately find their way to the stream. However, the zone of most intense interaction is within the floodplain and hyporheic zones, where subtle changes in oxygen levels can dramatically affect nutrient composition and bioavailability.

3.9.7 Microclimate

Although not well documented (O'Laughlin and Belt 1994), streamside vegetation can have a significant influence on local microclimates near the stream channel (FEMAT 1993). Chen (1991) reported that soil and air temperatures, relative wind speed, humidity, soil moisture, and solar radiation all changed with increasing distance from clear-cut edges in upslope forests of the western Cascades. Based on Chen's results, FEMAT (1993) concluded that loss of upland forests likely influences conditions within the riparian zone. FEMAT also suggested that riparian buffers necessary for maintaining riparian microclimates need to be wider than those for protecting other riparian functions (Figure 3-3).

3.9.8 Wildlife Habitat

Although riparian areas generally constitute only a small percentage of the total land area, they are extremely important habitats for wildlife. The attractiveness of riparian zones to wildlife likely reflects three attributes: the presence of water, which is essential to all life and generally scarce in the West (particularly east of the Cascade crest); local microclimatic conditions; and the more diverse plant assemblages found in riparian areas compared to surrounding uplands. The last characteristic derives from the dynamic nature of riparian zones, which typically leads to a mosaic of plant assemblages in different stages of ecological succession (Kauffman 1988). Brown (1985) reports that 87% of wildlife species in western Oregon and Washington use wetlands or riparian areas during some or all of their life cycle (FEMAT 1993). Thomas et al. (1979) found that 82% of all terrestrial vertebrates in the Great Basin of southeastern Oregon are either directly dependent on riparian zones or use riparian habitats more than any other habitat. Dependence of a majority of species on riparian zones has been demonstrated for all major vertebrate classes. Bury (1988) reported that 8 of 11 species of amphibians and 5 of 6 species of reptiles in Oregon either reside or breed in aquatic or riparian habitats. In northern California, approximately 50% of both reptiles and amphibians prefer riparian or aquatic habitats (Raedeke et al. 1988). Raedeke (1988) reviewed the published literature and found that 67% of native large mammals in the Pacific Northwest either depend on riparian areas or are more abundant in riparian areas than in surrounding uplands. Similar preferences for riparian habitat by small mammals, and especially bats, have also been documented (Cross 1988). Beschta et al. (1995) report that 55 species of birds in Oregon (approximately 46% of the total for which data were available) depend on or exhibit preferences for riparian habitats. For eastside ecosystems, the dependence of birds and other species on riparian zones is likely higher than for westside systems, where water and forests are more abundant.

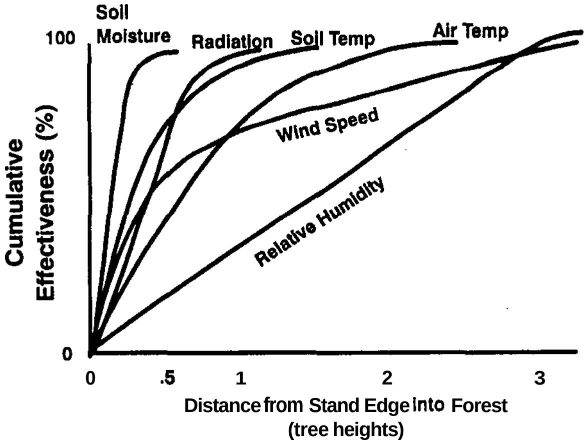


Figure 3-3. Riparian buffer effects on microclimate. From FEMAT (1993).

3.10 Implications for Salmonids

The above discussion highlights the highly complex array of physical and chemical processes that occur across the landscape, in the riparian zone adjacent to streams and rivers, and within the stream channel. Large-scale geomorphic and climatic processes have together shaped the landscape of the Pacific Northwest, exerting dominant control over channel gradient and configuration. Although these processes operate at scales of thousands to millions of years (Table 3-6), they are, nevertheless, important to consider in the development of salmonid conservation strategies. The current distribution of salmonids and other fish species in the Pacific Northwest is a direct consequence of tectonic activity dating back tens of millions of years that has both isolated and reconnected drainage basins through vertical lift and shifted subplates (McPhail and Lindsey 1986; Minckley et al. 1986). Similarly, climatic shifts and glaciation have alternately eliminated and stimulated reinvasion of fishes over significant portions of the Pacific Northwest landscape, as well as redistributed species into lower elevations or more southerly areas. Furthermore, the isolation of individual populations by geomorphic and glacial processes over time has allowed the evolution of unique stocks and species. Evolutionarily significant units (Waples 199lb) reflect the historical legacy wrought by geologic and climatic conditions over the millennia as well as adaptation to local environmental conditions that have prevailed since the last glacial period. Finally, long-term geomorphic and climatic processes together with hydrologic processes and vegetative cover, determine the rate at which nutrients, sediments, organic material, and water are transported from upslope areas into the stream channel. Consequently, the geomorphic and climatic setting determines the normal background rates of these processes, regulates the frequency and magnitude of natural episodic disturbances that reset and replenish streams, and govern the responses of specific watersheds to human perturbations.

Nested within this geomorphic and climatic context are a number of physical and chemical processes that further modify the landscape and that directly influence stream channel characteristics and water chemistry. These processes include surface erosion, landslides, floods, debris torrents, ice flows, droughts, beaver activity, and wildfire, and they operate at ecological times scales—generally from days to decades or centuries—regulating the input of sediment, nutrients, and organic material to the stream (Table ***3-6)*.** The riparian zone acts as the interface between terrestrial and aquatic ecosystems, moderating the effects of upslope processes as well as providing other critical functions (e.g., shading, bank stabilization, nutrient transformation, allochthonous inputs). Together, these processes determine macrohabitat characteristics, such as general channel morphology and pool-riffle sequences as well as microhabitat characteristics, such as depth, velocity, cover, temperature, and substrate.

The processes that influence salmonid habitats may be either cyclical in their occurrence (e.g., seasonal temperature, streamflow, and leaf-fall patterns), or episodic in nature (e.g., wildfires, landslides, floods, debris torrents). It is critical to recognize that these cycles or disturbances are fundamental and vital parts of ecosystem function, even though they may be temporarily disruptive of aquatic ecosystems. Studies of geomorphology and paleoecology indicate that disturbance is continual, sometimes across large areas, and often unpredictable. In eastside ecosystems the changes are most often associated with climatic changes that render vegetation more susceptible to disturbances such as fire and disease (Johnson et al. 1994). Eastside forests and rangelands have evolved with periodic disturbances, and when they do not receive them, they become increasingly unstable (Henjum et al. 1994; Johnson et al. 1994). If drought or fire do not alter these systems, then disease or insects will. Naturally occurring mass-soil movements and erosion introduce large woody debris, rock, gravel, and fine materials into stream channels, substantially modifying conditions for salmonids. Floods and debris torrents are dominant disturbances affecting westside stream systems (Swanston 1991) and may significantly alter channel morphology, scouring channels and creating debris jams and coarse sediment deposits that eventually produce important spawning and rearing areas for salmonids.

Salmonids have evolved not only to the general conditions that are typical of a watershed, but to the specific disturbance regimes found in that watershed. Human activities potentially modify disturbance regimes in three distinct ways: by increasing the frequency of disturbance events, by altering the magnitude of these events, and by affecting the response of the stream channel to disturbance events through modification of instream characteristics. Sediment delivery, for example, is essential to the development and maintenance of spawning gravels for salmonids. However, alteration in sediment composition, delivery rates, or fate can be damaging to salmonids, resulting in the degradation of spawning gravels and rearing habitats. Similarly, floods and droughts are important determinants of fish assemblage structure; however, increases in the frequency of these events may result in population declines, shifts in community structure, and decreases in biodiversity. The effects of human-caused alterations on salmonids and their habitats are discussed in greater detail in Chapter **6** of this document.

Table 3-6. Approximate ranges of recurrence of landscape and channel-forming processes and the effects of these events on stream habitats. Modified from Swanston (1991). Reproduced with permission from the publisher.

Event	Range of recurrence (years)	Channel changes	Habitat effects
Tectonics	1,000 - 1,000,000	Creation of new drainages; major channel changes including stream capture because of regional upwarping and faulting.	Subsidence in alluvial and coastal fills creating zones of deposition with increased fines; Steep erosive channels caused by upwarping leads to coarser sediments.
Climatic change	1,000 - 100,000	Major changes in channel direction; major changes in channel grade and configuration; valley broadening or downcutting; alteration of flow regime.	Changes in type and distribution of spawning gravels; changes in frequency and timing of disturbance events; shifts in species composition and diversity.
Volcanism	1,000 – 100,000	Local blocking and diversion of channel by mudflows and tephra; valley filling and widening; major changes in channel grade and configuration.	Changes in type and distribution of spawning gravels. Major inputs of sand and silt from tephra.
Slumps and earthflows	100 - 1,000	Low-level, long term contributions of sediment and large woody debris to stream channels; partial blockage of channel; local base level constriction below point of entry; shifts in channel configuration.	Siltation of spawning gravels; scour of channel below point of entry; accumulation of gravels behind obstructions; partial blockage of fish passage; local flooding and disturbance of side-channel rearing areas.
Wildfire	1 – 500	Increased sediment delivery to channels; increased large woody debris in channels; loss of riparian vegetation cover; decreased litterfall; increased channel flows; increased nutrient levels in streams.	Increased sedimentation of spawning and rearing habitat; increased summer temperatures; decreased winter temperatures; increased rearing and over-wintering habitat; decreased availability of fine woody debris; increased availability of food organisms.
Windthrow	10 – 100	Increased sediment delivery to channels; decreased litterfall; increased large woody debris in channel; loss of riparian cover.	Increased sedimentation of spawning and rearing habitat; increased summer temperatures; decreased winter temperatures; increased rearing and over-wintering habitat; decreased fine organic debris.
Insects and disease	10 – 100	Increased sediment delivery to channels: loss of riparian vegetation cover; increased large woody debris in channels; decreased litterfall.	Increased sedimentation of spawning and rearing habitat; increased summer temperatures; decreased winter temperatures; increased rearing and overwintering habitat.

Table 3-6. Approximate ranges of recurrence of landscape and channel-forming processes and the effects of these events on stream habitats. Modified from Swanston (1991). Reproduced with permission from the publisher.

Event	Range of recurrence (years)	Channel changes	Habitat effects
Activities of beavers	5–100	Channel damming; obstruction and redirection of channel flow; flooding of banks and side channels: ponding of streamflow; siltation of gravels behind dams.	Improved rearing and overwintering habitat; increased water volumes during low flows; slack-water and back-water refuge areas during floods; refuge from reduced habitat quality in adjoining areas; limitation on fish migration: elevated water temperatures; local reductions in dissolved oxygen.
Debris avalanches and debris torrents	5–100	Large, short-term increases in sediment and large woody debris contributions to channel; channel scour; large-scale movement and redistribution of bed-load gravels and large woody debris; damming and obstruction of channels; accelerated channel bank erosion and undercutting; alteration of channel shape by flow obstruction; flooding.	Changes in pool to riffle ratio; shifting of spawning gravels: siltation of spawning gravels; disturbance of side-channel rearing areas; blockage of fish access; filling and scouring of pools and riffles; formation of new rearing and overwintering habitat.
Major storms; floods; rain-on-snow events	1.0–10	Increased movement of sediment and woody debris to channels; flood flows; local channel scour; movement and redistribution of coarse sediments; flushing of fine sediments: movement and redistribution of large woody debris.	Changes in pool to riffle ratio; shifting of spawning gravels; increased large woody debris jams; siltation of spawning gravels; disturbance of side-channel rearing areas; increased rearing and overwintering habitat; local blockage of fish access: filling and scouring of pools and riffles.
Seasonal precipitation and discharge: moderate storms: freezing and ice formation	0.1 – 1.0	Increased flow to bank-full width; moderate channel erosion; high base-flow erosion: increased mobility of in-channel sediment and debris; local damming and flooding; sediment transport by anchor ice; gouging of channel bed; reduced winter flows.	Changes in pool to riffle ratio; siltation of spawning gravels; increased channel area; increased access to spawning sites; flooding of side-channel areas; amelioration of temperatures at high flows; decreased temperatures during freezing; dewatering of gravels during freezing; gravel disturbance by gouging and anchor ice.
Daily to weekly precipitation and discharge	0.01 – 0.1	Channel width and depth; movement **and** deposition of fine woody debris; fine sediment transport and deposition.	Minor siltation of spawning gravels; minor variation in spawning and rearing habitat; increased temperature during summer low flows.

4 Biological Processes and Concepts

The physical and chemical environment of aquatic ecosystems forms the template upon which biological systems at all levels are organized (Southwood 1977; Poff and Ward 1990). The spatial and temporal patterns of water quantity and velocity, temperature, substrate, and dissolved materials influence the physiology and behavior of organisms, the dynamics and evolution of populations and metapopulations, and the trophic structure and diversity of aquatic communities. Modification of physical and chemical attributes of water bodies through land-use and water-use practices, and direct alteration of specific biological components of these systems, can result in changes to individual organisms, populations, and communities. In this section, we briefly review some fundamental biological processes that occur in aquatic ecosystems, focusing on those processes that are likely to be affected by modifications to physical and chemical habitat characteristics.

4.1 Organism Level

The survival of salmonids in the wild depend on their ability to carry out basic biological and physiological functions including feeding, growth, reproduction, respiration, hydromineral balance, smoltification (anadromous forms), and migration. The fate of populations and the outcome of higher-level biological interactions—competition, predation, and disease—ultimately depend on the performance of individuals in obtaining food, defending space, maintaining physiological health, or otherwise coping with their ecological circumstances. Characteristics of the physical and chemical environment of fish, particularly water temperature, regulate the rates at which these processes occur. A detailed discussion of the complex interactions between fish and their environments is beyond the scope of this document; however, a brief review of the fundamental biological processes is essential to understanding how habitat modifications may affect salmonids.

4.1.I Feeding and Growth

Juvenile salmonids are generally opportunistic in their feeding habits while in freshwater, primarily consuming drifting aquatic or terrestrial invertebrates in streams, and macroinvertebrates and zooplankton in lakes and estuaries. Bull trout especially, as well as resident rainbow and cutthroat trout, may feed on other fishes and amphibians during their adult stages, particularly in systems where they attain large sizes. A summary of specific dietary items for anadromous and resident salmonids can be found in Meehan and Bjornn (1991); a more detailed examination of dietary habits of Pacific salmon can be found in Groot and Margolis (1991).

Environmental conditions influence the demand for food, the amount and type of prey available to salmonids, the ability of fish to capture prey, and the costs of obtaining food. Ingestion rates of fishes generally increase with increasing temperature, except when temperatures exceed the thermal optimum for the species (Brett 1971). Low levels of dissolved oxygen may also lead to suppression of appetite in salmonids (Jobling 1993). Increased levels of sediment may alter substrate composition, filling substrate interstices, and thereby affecting the total abundance and composition of invertebrate prey. Similarly, reductions in fine and coarse organic litter inputs can both reduce the food base and alter habitat structure for prey organisms. Turbidity in streams may reduce light penetration, decreasing the reactive distance of salmonids to prey and limiting production of benthic algae. Nutrient availability also affects total food availability by controlling primary production. For stream-dwelling salmonids, the energetic costs associated with acquiring food depend on current velocities at holding and feeding stations. Many salmonids seek out areas of slow water velocity immediately adjacent to faster waters, presumably because these areas provide greater food per unit of energy expended in maintaining position (Smith and Li 1983; Fausch 1984). Heterogeneity of velocity, therefore, creates microhabitats that are energetically favorable. All of these factors can be affected by human alterations of habitat or watershed processes, fundamentally affecting the ability of individuals to satisfy their food intake requirements.

Once food energy is consumed, it is used in a variety of metabolic processes. These include respiratory and circulatory processes that deliver oxygen to various tissues, maintenance of cells, digestion of food, assimilation and storage of nutrients, and various muscular activities (e.g., swimming and other behaviors). After satisfying these demands, surplus energy that is not excreted as waste can be devoted to growth of body and reproductive tissues. A number of environmental variables influence the growth rate of salmonids. In general, growth rates increase with increasing temperature up to a thermal optimum, above which reductions in appetite and increasing metabolic demands combine to reduce the growth rate. Growth rates of salmonids, as well as food conversion efficiency, may also be reduced when dissolved oxygen levels below 5–8 mg/L or 60%–70% of saturation (Jobling 1994). Other chemical factors that influence growth rate include ammonia and salinity (Moyle and Cech 1982), as well as various other pollutants. Because water velocity determines metabolic demands of fish, it indirectly determines how much energy is available for anabolic processes. Thus, human-caused changes in water quality, natural flow regimes, or hydraulic characteristics all may inhibit growth and development of salmonids.

4.1.2 Reproduction and Embryological Development

Energy reserves of salmonids must be sufficient to allow for gamete production after growth and metabolic costs are incurred. Anadromous salmonids have particularly high energy requirements because they must have sufficient reserves to undergo lengthy migrations and negotiate barriers in order to reach the ocean and then return to their spawning tributaries. Modifications of temperature, water quality, streamflow, and physical structure all affect how much energy can be devoted to reproductive output. The development of embryos and alevins in the gravel is affected by several environmental factors. Water temperature greatly influences times to hatching and emergence for Pacific salmonids. Development time decreases in an asymptotic fashion with increasing incubation temperatures with the rate of change in development time relative to temperature increase being greatest at the low end of the tolerable temperature range (Beacham and Murray 1990). Consequently, small increases in temperature at the low end of the range can substantially alter the time of hatching and emergence of salmonids. Early emergence because of warming of water temperatures may increase exposure of fry to high-flow events and alter the natural synchrony between emergence and predator cycles or prey cycles. Scrivener (1988) found that chum salmon in Carnation Creek emerged and migrated to sea 4 to 6 weeks earlier after logging compared with prelogging years in response to water temperature increases. In a companion study, Holtby (1988) reported that coho salmon emerged up to 6 weeks early in response to logging. Temperatures may influence the size of emerging fry. For example, coho salmon reared at 4°C were larger than those reared at warmer temperatures (Beacham and Murray 1990). In contrast, fry of pink salmon tended to be larger when reared at 8°C than when reared at 4°C.

Dissolved oxygen concentrations in redds also influence the survival and development rate of embryos and alevins, as well as the size of emerging fry (Warren 1971). Streamflow may regulate the flow of water through redds and hence the levels of dissolved oxygen. Environmental changes, such as siltation or altered flow regimes, that reduce the flow of water can thus adversely affect embryo and alevin development and survival. Silt concentration in gravels may also impede emergence of fry. Phillips et al. (1966) found that emergence of coho fry decreased as the percentage of fine sediments in the gravel increased, presumably because of reduced oxygen content and increased difficulty of fry in reaching the surface. Increased frequency of high scouring flows or debris torrents, which are associated with disturbed catchments (Swanston 1991), may further affect egg and alevin survival.

4.1.3 Respiration

Most of the energy used by salmonids to swim, locate food, grow, and reproduce is provided through metabolic processes that require oxygen. Because water contains only about 3.3% of the amount of oxygen contained in air, the efficient extraction of oxygen is critical to survival. Fish, and salmonids in particular, have evolved elaborate gill structures that facilitate the uptake of oxygen for delivery to other parts of the body. Environmental conditions can have a significant influence on the oxygen demands of fish, the amount of oxygen present in water, and the ability of fish to take up that oxygen. In general, the oxygen demands increase with increasing temperature, although oxygen consumption may decrease as temperatures approach lethal levels, particularly at high levels of activity (Brett 1971). In contrast, dissolved oxygen levels in saturated water are inversely proportional to temperature with water at 5°C holding approximately 30% more oxygen than water at 20°C. Oxygen demand is also influenced by water velocity, which determines the swimming speed required of salmonids to maintain their position in the current. High levels of suspended solids in water may influence respiration by abrading or clogging gill surfaces (Warren 1971). Similarly, pollutants can cause mucous secretions to coat gill surfaces, inhibiting the exchange of oxygen. Excessive amounts of algae and easily decomposable organic material in water increases plant and microbial oxygen demand, thereby decreasing dissolved oxygen concentration. Low levels of dissolved oxygen, in turn, impede the ability of hemoglobin within the blood to bind with oxygen, effectively reducing the delivery of oxygen to body tissues (Moyle and Cech 1982). On the other hand, gas supersaturation from dam spills and intense algal photosynthesis can create gas bubbles in fish gills and tissue, resulting in decreased respiratory efficiency, disease, or death. All of these factors can influence the ability of fish to satisfy their oxygen demands.

4.1.4 Smolfification

The transition from fresh to salt water marks a critical phase in the life history of anadromous salmonids. Emigration to the ocean is preceded by rapid physiological, morphological, and behavioral transformations that preadapt fish for life in salt water and initiate their downstream movement (Folmar and Dickhoff 1980; Wedemeyer et al. 1980; Groot 1982). Once at sea, newly arrived smolts must acclimate to a markedly different set of ecological circumstances, including new food resources, new predators, and a substantially different physical environment. Much of the total ocean mortality incurred by salmon smolts is believed to occur during this period of early ocean life (Manzer and Shepard 1962; Matthews and Buckley 1976; Walters et al. 1978; Fisher and Pearcy 1988; Pearcy 1992). Consequently the timing of ocean entry is likely adaptive to maximize survival or growth (Miller and Brannon 1981; Riddell and Leggett 1981; Murphy et al. 1988; Beacham and Murray 1990).

Because development and growth are highly influenced by water temperatures, modifications to thermal regimes can potentially alter the time of smoltification (reviewed in Wedemeyer et al. 1980; Hoar 1988). Similarly, temperature and streamflow patterns may be important cues for releasing migratory behavior (Hoar 1988). Consequently, alterations in normal hydrologic and thermal patterns may trigger movement into the ocean at times that are less favorable for growth and survival. The parr-smolt transformation may also be affected by exposure to contaminants, including heavy metals, which alter enzymatic systems involved in osmoregulation and may inhibit migratory behavior (Wedemeyer et al. 1980). Structural alterations that hinder salmonids during the smolt transformation include loss of large woody debris and habitat complexity in streams and estuaries, which reduces cover and food supplies during this critical period.

4.1.5 Summary

A useful way of summarizing the effects of environmental factors on individual fish is through a simple energy budget. Food energy that is ingested by fish (I) has several potential fates. It is either expended during metabolic processes (M), deposited as new somatic (body) or reproductive tissue (G), or excreted as waste products (E) (Jobling 1993). Thus the energy balance can be expressed as

$$I = M + G + E \ .$$

Environmental conditions influence all aspects of a fish's energy budget. Temperature, in particular, has pervasive effects on bioenergetic pathways, affecting appetite, digestion rate, standard and active metabolic rates, and food conversion efficiency (i.e., the proportion of food energy absorbed by the fish). Because the energetic costs of swimming increase exponentially with increasing speed (Jobling 1993), water velocity determines how much energy is expended in maintaining position and obtaining food. Dissolved oxygen concentrations also affect food consumption and metabolic processes (Warren 1971), as do various chemical pollutants. These and other environmental factors interact to determine the amount of energy expended on metabolic processes, and hence determine the energy left over for growth and reproduction. When changes in environmental conditions reduce the amount of food available or alter the efficiency with which food is captured and assimilated, the performance of individual fish declines. This reduction in performance, in turn, affects the outcome of higher-level interactions including competitive, predator-prey, and disease-host relationships (see Section 4.3).

4.2 Population Level

Salmonid populations are noted for their complex life cycles, diverse life histories, and tendency to form locally adapted stocks. The interaction among various subpopulations (metapopulation dynamics) has important implications for conservation.

4.2.1 Generalized Life Cycle

The life cycle of anadromous salmonids consists of several distinct phases, at least three of which involve significant shifts in habitat. Adult salmon migrate from the ocean into their natal stream to spawn. Females construct a "redd" in the stream gravel into which eggs are deposited, fertilized by males, and subsequently covered with gravel. All adult salmon die after spawning, usually within a few weeks. Females will typically spend one to three weeks guarding the redd site before dying, whereas males may seek out and spawn other females. The fertilized embryos develop for a period of one to several months, depending on temperature and dissolved oxygen availability, before hatching occurs. The emergent "alevins" remain in the gravel, nourished by a yolk sac, for another few weeks to a month or more. Once yolk-sac absorption is complete, the fry emerge from the gravel and begin actively feeding on drifting material. The period of freshwater rearing lasts from a few days to several years, depending on the species, after which juveniles undergo smoltification. Smolts migrate to the ocean, where the majority of growth occurs, before returning to spawn as adults, completing the cycle (Figure 4-1).

The life cycles of the anadromous trout and char differ from those of salmon in that some adults may survive after spawning, migrate back to the ocean, and return to spawn a second or third time. Resident salmonids, including kokanee salmon (i.e. landlocked sockeye salmon), bull trout, cutthroat trout, rainbow trout, and mountain whitefish, do not have an oceanic phase but commonly undergo substantial migrations to and from rearing areas in lakes or larger rivers. With the exception of kokanee salmon, which die after spawning, the resident forms usually spawn multiple times over their lifetimes.

4.2.2 Life History

Although all anadromous salmonids share the general life cycle discussed above, substantial differences exist in the period of time that the different species spend in freshwater and marine environments (Table 4-1), and the types of habitat they use for spawning and rearing. In addition, a high degree of variation in life histories can exist within each species.

Life-History Patterns

Extensive reviews of the life histories and general habitat preferences of trout, char, and Pacific salmon can be found in Groot and Margolis (1991) and Meehan and Bjornn (1991) from which much of the information below was taken. Pink and chum salmon typically spawn in coastal streams not far from tidewater—chum occasionally within the tidal zone—and have the shortest freshwater phase, entering the ocean soon after they emerge from the gravel. Almost without exception, pink salmon mature at 2 years of age, at which time they return to freshwater to spawn. Chum salmon are more variable, spending from 2 to 5 years in the ocean before returning to their natal area to spawn. Coho salmon generally spawn in small, low-gradient streams or stream reaches in both coastal and interior systems. Juveniles typically spend from 1 to 3 years in freshwater; however, in the southern portion of their range (including Washington, Oregon, and California) most fish migrate to sea after just one year. Adults return to spawn after approximately 18 months at sea, although "jack" males may return after only six months in the ocean (Sandercock 1991). The life histories of sockeye and chinook salmon are more variable. Sockeye salmon most often spawn in the inlet or outlet streams of lakes. Shortly after emergence, sockeye fry migrate into these lakes, where they reside for 1 to 3 years. Juveniles then migrate to the ocean, where they spend 2 to 3 years. Chinook salmon generally spawn in small to medium-sized rivers, but may also spawn in large river systems such as the mainstem Columbia. Chinook salmon display two dominant life-history types, an ocean type that is typical of fall-run stocks and a stream type that is characteristic of spring-run fish. Those exhibiting the ocean-type life history usually spend only a few months in freshwater before migrating to sea. Stream-type fish spend 1 to 2 years in freshwater. Both ocean- and stream-type fish can reside anywhere from 2 to 5 years in the ocean, although jacks may spend less than a year at sea before returning to spawn. Within any given population, multiple life-history patterns may be observed. Based on time of freshwater and estuarine residence, Reimers (1973) identified five distinct life-history patterns for fall chinook salmon in the Sixes River, Oregon.

The anadromous trout and char, including steelhead and cutthroat trout, and Dolly Varden, exhibit considerable life-history variation as well (Table 4-1). Steelhead trout tend to spawn in small streams and favor relatively high-gradient reaches. Freshwater residence can last from 1 to 4 years, while ocean residence ranges from a few months ("half-pound") males to 4 years. Although most adult steelhead die after spawning, up to 30% may live to return to the ocean and spawn again in subsequent years, particularly in coastal streams where the spawning migrations are fairly short (Meehan and Bjornn 1991). Consequently, the number of potential life-history types is large. Anadromous cutthroat trout most commonly spawn in small headwater streams and spend 2 to 4 years in freshwater before migrating to the ocean during the spring, where they generally remain until the next fall. As with steelhead trout, some adults may live after spawning, migrate back to the ocean, and return a second or third time. Dolly Varden spawn in coastal streams and exhibit complex life-history patterns. Juveniles typically rear in higher-velocity habitats for several years (Meehan and Bjornn 1991). After smoltification, Dolly Varden enter the ocean, but may repeatedly return to freshwater habitats during the winter months to rear in lakes, sometimes away from their natal areas. Thus, it is difficult to generalize about the periods of freshwater and marine residence for Dolly Varden.

Resident trout, char, and whitefish spend their entire lives in freshwater; however, life-history patterns may still be quite diverse (Table 4-1). Varley and Gresswell (1988) identified four principal life-history patterns for Yellowstone Lake cutthroat trout: fluvial populations that remain in their natal streams throughout their lives, fluvial-adfluvial populations that reside in larger rivers but spawn in small tributaries, lacustrine-adfluvial populations that

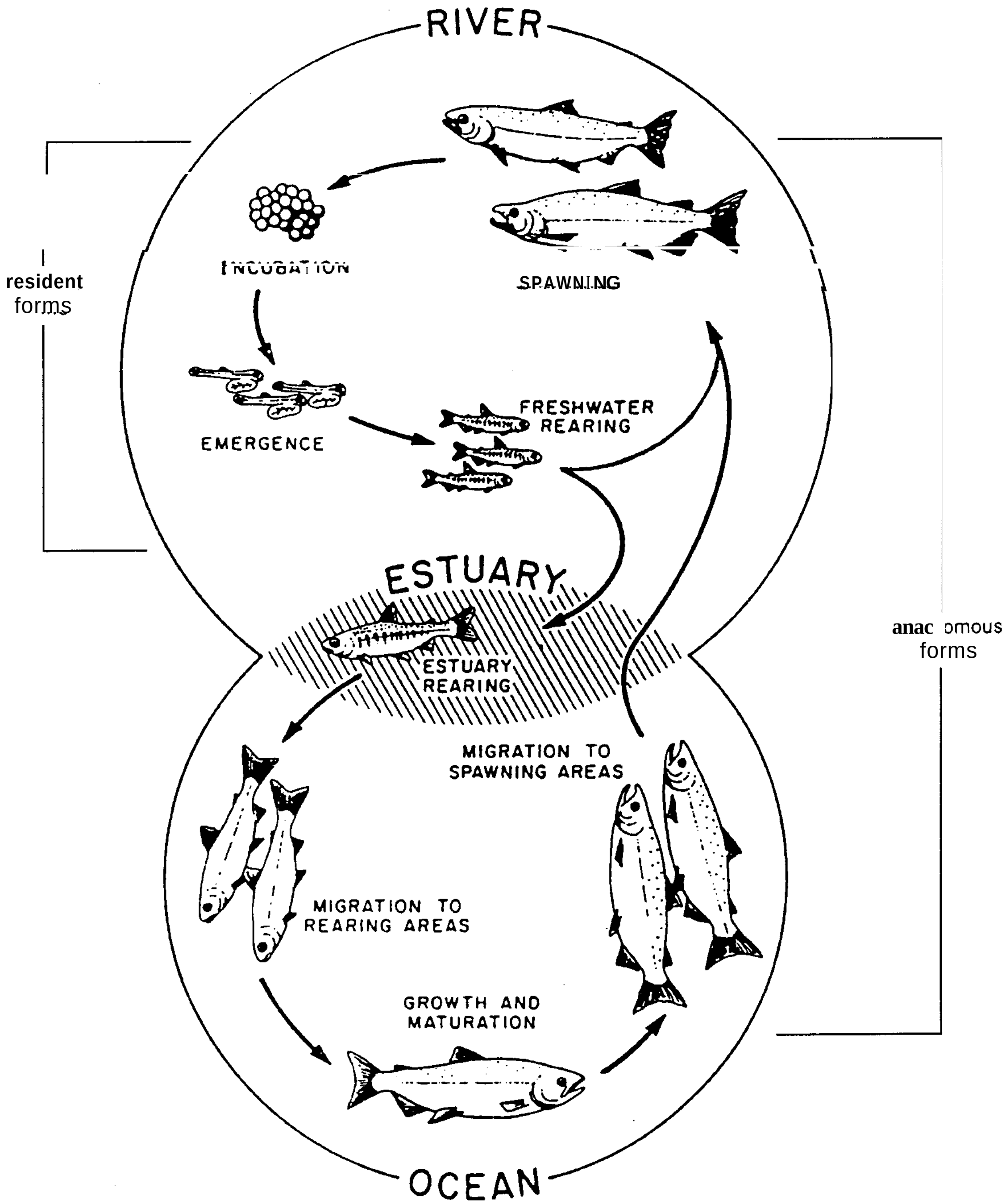

RIVER
resident forms
INCUBATION
SPAWNING
EMERGENCE
FRESHWATER REARING
anadromous forms
ESTUARY
ESTUARY REARING
MIGRATION TO SPAWNING AREAS
MIGRATION TO REARING AREAS
GROWTH AND MATURATION
OCEAN

Figure **4-1.** Generalized salmonid life cycle, showing freshwater and ocean components. Modified from Nicholas and Hankin (1988). Reproduced with permission from the authors.

Table 4-1. Life histories of Pacific salmonids.*

Species	Spawning migration	Spawning period	Spawning area	Life history	Most common age at maturity†
ANADROMOUS SALMON					
Chum salmon	Summer to winter	Summer to winter	Usually near tidewater	Fry go directly to sea. 2 – 5 years ocean.	4_1
Pink salmon	Late summer to early fall	Late summer to early fall	Usually near tidewater	Fry go directly to sea. 2 years ocean.	2_1
Sockeye salmon	Spring to fall	Late summer to fall	Tributaries of lakes	1 – 3 years lake. 2 – 3 years ocean.	4_2 5_3
Coho salmon	Summer to fall	Fall to early winter	Small headwater streams	1 – 3 years FW.‡ 6 months jack. } 18 months adult } ocean.	3_2
Chinook salmon	Spring to fall	Summer to early winter	Large rivers	3 months – 2 years FW. 2 – 5 years ocean.	4_1 (Ocean) 5_1 (Stream)
ANADROMOUS TROUT AND CHAR					
Steelhead trout	Summer to winter	Late winter to spring	Small headwater streams	2 – 3 years FW. 1 – 3 years ocean. Repeat spawners.	mature 4 – 5
Searun cutthroat trout	Fall to winter	Late winter to early spring	Small headwater streams	2 – 4 years FW. 2 – 5 months ocean. Repeat spawners.	mature 3 – 4
Dolly Varden	Late summer to fall	Fall	Main channels on rivers	2 – 4 years FW. 2 – 4 years ocean. Repeat spawners.	mature 5 – 6 die 6 – 7
RESIDENT SPECIES					
Kokanee salmon	Late summer to fall	Late summer to fall	Tributaries of lakes, lakeshores	Juveniles migrate to lakes to reside.	3 – 4
Rainbow trout	Spring	Spring	Small headwater streams	Fluvial, adfluvial, lacustrine-adfluvial life histories. Variable residence in natal streams, rivers, & lakes.	2 – 3
Cutthroat trout	Spring	Spring to early summer	Small headwater streams	Fluvial, adfluvial, lacustrine-adfluvial life histories. Variable residence in natal streams, rivers, & lakes.	3 – 4
Bull trout	Fall	Fall	Large streams with ground water infiltration	Juveniles migrate from tributaries to lakes or larger streams at about 2 years, highly variable.	4 – 9
Mountain whitefish	Fall	Fall	Mid-sized streams, lakes	Reside in streams and lakes.	3 – 4

* Data from Groot and Margolis (1991); Meehan and Bjornn (1991); Pratt (1992); Behnke (1992); and Moyle (1976).

† Gilbert-Rich age designation in years.

‡ FW = freshwater.

reside in lakes and spawn in tributary streams, and allucustrine populations that reside in lakes and migrate down outlet streams to spawn. Rainbow trout may spawn in streams, in lake inlets or outlets, or in lake springs, and rear in streams or lakes (Behnke 1992). Bull trout reside in a variety of freshwater habitats including small streams, large rivers, and lakes or reservoirs (Meehan and Bjornn 1991). Some populations spend their entire lives in cold headwater streams. In other populations, juveniles spend from 2 to 4 years in their natal stream before migrating into lakes or reservoirs, where they reside for another 2 to 4 years before returning to their natal stream to spawn. Mountain whitefish spawn in streams and rivers and reside there throughout their lives although substantial migrations from larger rivers into smaller spawning tributaries are common.

Implications of Life-History Diversity for Salmonid Conservation

The remarkable diversity of life histories exhibited by Pacific Northwest salmonids reflects adaptation to a wide array of habitats. As a group, the salmonids inhabit streams ranging from mountain headwaters to large lowland rivers, in regions varying from arid and semiarid shrublands to temperate rainforests. Reproduction may occur in streams, lakes, or intertidal sloughs; rearing of juveniles occurs in streams and lakes for some species and in estuaries and oceans for others (Table 4-2). In any particular habitat, spatial and temporal differences in micro- and macrohabitat utilization permit the coexistence of ecologically similar species (Everest et al. 1985). Within species, life-history diversity allows salmonids to fully utilize available freshwater, estuarine, and ocean environments. Species that occupy several habitat types, or that have multiple freshwater and marine residence times, effectively spread ecological risk (sensu Den Boer 1968) such that the impacts of environmental fluctuation on populations are distributed through time and space. Consequently, species are likely to differ in their response to human-caused perturbation. The diversity among species and by life stage indicates that most accessible freshwater habitats are used year round if environmental conditions are suitable (Table 4-3).

Life-history diversity should be considered in the development of salmonid conservation strategies and local enhancement measures (Carl and Healey 1984; Lichatowich et al. 1995). The historically high abundance of salmonids in the Pacific Northwest was due in part to the diversity of life-history patterns exhibited by the various species. Habitat simplification through land-use and water-use practices has effectively simplified this diversity in life-history organization. In the Columbia River, for example, salmonids of various species and life stages were present in the mainstem year round. Because of alteration of temperatures and flow regimes, the temporal usage of the mainstem and major tributaries is now far more restricted. Historical records indicate that the Yakima River once supported **six** life-history types of spring chinook salmon, two of which reared in warmer, low-elevation, mainstem reaches. Today, because irrigation withdrawals have reduced flows and increased temperatures, the population consists only life-history types that rear in upper tributaries—the life-history types that utilized the lower mainstem for rearing have been eliminated (Lichatowich et al. 1995). Restoration of such populations to a harvestable level will require restoration of habitat conditions suitable for all life-history types of chinook salmon. Differences in life histories also affect the response of salmonids to harvest. Salmon that spend several years at sea before maturing are more vulnerable to troll fisheries than those that spend only a year at sea (see Section 6.11).

4.2.3 Stock Concept and Local Adaptation

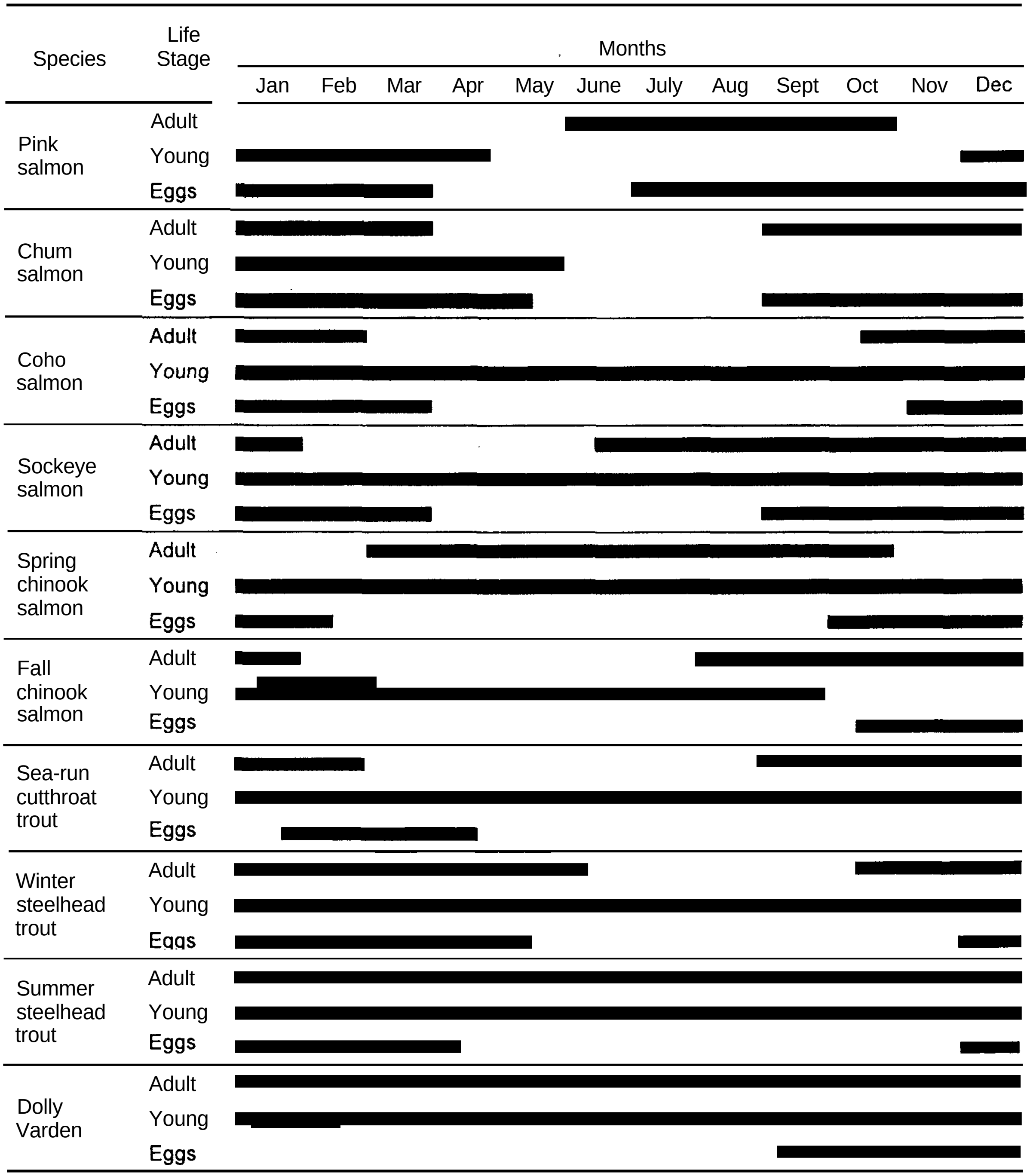
Among the most remarkable characteristics of anadromous salmonid species is their tendency to return to their natal stream to spawn during a particular season often after ocean migrations of a thousand miles or more. Although the strong homing tendency of salmonids is most conspicuous in anadromous species, it may be common in resident populations as well. Lake-dwelling populations of cutthroat and bull trout that spawn in tributaries have also been shown to return to their natal stream to spawn with low rates of straying (Pratt 1992; Gresswell et al. 1994), and it is likely that stream-dwelling residents also display some fidelity to their natal area. As a consequence of homing, salmonid species typically comprise numerous local populations or "stocks" that are to varying degrees reproductively isolated from other such populations. Ricker (1972) defined a stock as "the fish spawning in a particular lake or stream (or portion of it) at a particular season [that] to a substantial degree do not interbreed with any group spawning in a different place, or in the same place at a different season."

The homing and resultant reproductive isolation of stocks provide a mechanism by which local populations become uniquely adapted to the specific suite of environmental conditions encountered during their life histories. Ricker, in his classic 1972 paper that formalized this concept, catalogued dozens of examples of local variation in morphological, behavioral, and life-history traits and provided evidence that many of these traits are to some degree heritable. For a trait to be considered adaptive, it must not only be differentially expressed, but it must

Table 4-2. Variation in life histories of Pacific salmonids. Modified from Everest et al. (1985).

Species (race)	Life histories	Spawns in			Rears in			
		Lakes	Streams	Intertidal	Lakes	Streams	Estuaries	Ocean
Pink salmon	Anadromous		X			X	X	X
	Anadromous		X					X
	Anadromous			X			X	X
Chum salmon	Anadromous		X			X	X	X
	Anadromous		X			X		X
	Anadromous		X					X
	Anadromous			X			X	X
Coho salmon	Anadromous		X			X	X	X
	Anadromous		X			X		X
Sockeye salmon	Anadromous		X		X			X
	Anadromous	X			X			X
Kokanee salmon	Resident		X		X			
	Resident	X			X			
Chinook salmon (spring)	Anadromous		X			X	X	X
	Anadromous		X			X		X
Chinook Salmon (fall)	Anadromous		X			X	X	X
	Anadromous		X			X		X
Cutthroat trout	Resident		X			X		
	Resident		X		X			
Cutthroat trout (searun)	Anadromous		X			X	X	X
	Anadromous		X			X		X
Rainbow trout	Resident		X			X		
	Resident		X		X			
	Resident	X			X			
Steelhead trout	Anadromous		X			X		X
Bull trout	Resident		X			X		
	Resident		X		X			
Dolly Varden	Anadromous		X			X	X	X
	Anadromous		X			X		X
	Anadromous		X		X			X
Mountain whitefish	Resident		X			X		
	Resident	X			X			

Table 4-3. Seasonal occurrence of adult, embryonic, and juvenile anadromous salmonids in freshwaters of western Oregon and Washington. From Everest et al. (1985).



confer some advantage to the individuals exhibiting that trait. More formally, Taylor (1991) defined local adaptation as "a process that increases the frequency of traits within a population that enhance the survival or reproductive success of individuals expressing such traits." He identified many examples of variation in morphological, behavioral, developmental, biochemical, physiological, and life-history traits in the family Salmonidae that are both heritable and believed to be adapted to local conditions. Results from his extensive review are summarized in Table 4-4.

Despite the fact that the stock concept is generally credited to Ricker, the implications of stock formation and local adaptation in conservation of salmonid species have long been recognized. Rich (1939) proposed that conservation of a species that is made up of numerous, isolated, self-perpetuating units depends on conserving each constituent part. While Rich argued that local adaptation was not necessary for stocks to be the appropriate unit of management, the recognition that stocks do differ in heritable traits and that these differences are a consequence of differential selection serves to strengthen the argument for conserving individual salmonid stocks. The loss of local stocks changes the genetic composition and reduces the genetic variability of the species as a whole (Nehlsen et al. 1991), reducing its ability to respond to environmental change.

From Table 4-4 it is evident that many traits of salmonids are adaptations to environmental conditions that may be significantly altered by human activities. In the wake of rapid and extensive anthropogenic change, traits that were once adaptive may be rendered maladaptive. For example, the timing of spawning, emergence, and smoltification of salmonids are clearly linked to stream temperature regimes as are development rates of eggs and juveniles. Warming of stream temperatures through loss of riparian canopy, releasing water from reservoirs, or using irrigation practices can advance development or alter the timing of life-history events and potentially disrupt natural synchronies in biological cycles that have evolved over thousands of years. Alteration of temperatures may also affect embryo and alevin survival as well as enzyme activity in populations that are specifically adapted to warm or cool environments. Thus, small changes in temperature may prove ecologically damaging even though such changes would produce no evidence of acute or chronic physiological stress. Other characteristics, including body morphology, agonistic and rheotactic behavior, and the timing of smolt and adult migrations, are tied to streamflow. Changes in the timing or magnitude of flows because of hydroelectric operations, agricultural diversions, or disruption of hydrologic processes from forest and range practices may effect these characteristics of fish. In this context, the ability of species-specific (versus stock-specific) criteria for water quality, instream flows, and other habitat attributes to adequately protect individual salmonid stocks should be re-evaluated. These stock differences are one reason that hatcheries threaten biological diversity of wild stocks (see Section 6.12).

4.2.4 Mefapopulation Dynamics

The stock concept focuses on the reproductive isolation and subsequent adaptation of local populations to the particular environments that they inhabit. Metapopulation theory is concerned with the behavior of groups of populations, or "metapopulations," that interact via individuals moving among populations through the processes of dispersal or straying (Hanski and Gilpin 1991). The term "metapopulation dynamics" thus describes the long-term behavior of a metapopulation over time.

Early theoretical work on metapopulations focused on extinction and recolonization rates of subpopulations making up a metapopulation (Levins 1969). Local populations within a metapopulation periodically go extinct as a result of natural disturbances or fluctuations in environmental conditions, leaving vacant habitat patches that may subsequently be recolonized by individuals from other populations (Hanski and Gilpin 1991). Under the model of Levins (1969), each subpopulation within the metapopulation has an equal probability of extinction; thus metapopulation persistence requires that, among local populations, the recolonization rate must exceed the extinction rate (Hanski 1991).

More recently, metapopulation models have been proposed that assume various subpopulations play different roles in metapopulation dynamics (Harrison 1991; Hanski 1991). One such model, the core-satellite model, describes a metapopulation where a larger core population gives rise through dispersal to numerous satellite populations (Harrison 1991). In these circumstances, metapopulation persistence depends on the existence of a few extinction-resistant source populations serving as sources of colonists for extinction-prone satellite populations. In a dynamic environment, the role of various subpopulations may change through time; source populations may become sinks and vice versa. Even where local extinction does not occur, depression of populations may influence genetic interactions among populations constituting the metapopulation.

While discussion of metapopulation dynamics of anadromous and resident salmonids is largely absent from the literature (but see Li et al. 1995); a number of principles from metapopulation theory relate to salmonid conservation. Evidence from other

Table 4-4. Examples of local variation in traits of salmonids and their presumed adaptive advantages.*

Trait	Adaptive Advantage	Species (Life cycle phase)†	Source
MORPHOLOGY AND MERISTICS			
Streamlined body and larger fins	Streamlined body and larger fin size adapted for higher water velocities.	Coho salmon **(J)** Atlantic salmon (J) Pink salmon **(A)** Chum salmon (A)	Taylor and McPhail (1985) Riddell and Leggett (1981) Beacham (1985); Beacham et al. (1988b) Beacham (1984); Beacham and Murray (1987); Beacham et al. (1988a)
Jaw size and shape	Adapted to local differences in food particle size.	Arctic char	Barbour (1984) Skulason et al. (1989)
Gill raker number and length	Adapted to local differences in food particle size.	Lake whitefish	Ihssen et *al.* (1981) Lindsey (1981)
BEHAVIOR			
Direction of fry migration	Emerging fry migrate in direction of rearing lakes.	Sockeye salmon (F) Rainbow trout (F)	Brannon (1972); Raleigh (1971) Raleigh (1971); Kelso et al. (1981)
Compass orientation *of* emerging fry	Local differences in orientation facilitate migration to feeding areas.	Sockeye salmon (F)	Quinn (1982, 1985)
Rheotactic behavior	Adapted to local differences in optimal timing of downstream migration.	Chinook salmon ***(S)***	Taylor (1990b)
Homing accuracy	Differences in straying rates potentially reflect differences in environmental stability.	Pink salmon (A)	Bams (1976)
Migratory behavior	Adapted to local differences in predation pressure, local feeding areas, or hydrologic characteristics.	Brown trout (A)	Svardson and Fagerstrom (1982)
Migration timing	Differences between fall and spring races reflect seasonal variation in accessibility to spawning streams.	Chinook salmon **(A)**	Belding and Kitson (1934); Smith (1969)
Spawning timing	Differences in spawning timing reflect temperature differences in streams.	Pink salmon **(A)** Chinook salmon **(A)** Sockeye salmon (A)	Sheridan (1962); Royce (1962) Burger et *al.* (1985) Brannon (1987)
Agonistic behavior	Reduced level of agonistic behavior in "ocean type" juveniles compared with "stream types" that establish territories.	Chinook salmon (J)	Taylor (1988, 1990b)
	Lower levels of agonistic behavior in populations with high predation; displays **may** increase **risk.**	Coho salmon **(J)**	Rosenau and McPhail (1987)
	Lower levels of agonistic behavior for fish in lakes or other slow\moving habitats. Higher levels in stream-dwelling fish, where territorial defense is advantageous.	Coho salmon **(J)**	Grant and Noakes (1988) Swain and Holtby (1989)

Table **4-4.** Examples of local variation in traits of salmonids and their presumed adaptive advantages.*

Trait	Adaptive Advantage	Species (Life cycle phase)†	Source
DEVELOPMENTAL BIOLOGY			
Embryo/alevin survival	Survival of embryos/alevins from populations native to coldwater environments greater at low temperatures than for populations from warmwater environments (and vice versa).	Chum salmon (E) Sockeye salmon (E) Pink salmon (E)	Tallman (1986) Beacham and Murray (1989) Beacham (1988); Beacham and Murray (1988)
Developmental rate	Faster development in late spawning stocks may facilitate synchronous emergence with fry of early spawners. Synchrony adaptive for predator swamping or narrow window of favorable oceanic conditions.	Chum salmon (E)	Tallman (1986)
BIOCHEMICAL TRAITS			
Lactate dehydrogenase	Temperature-dependent selection of certain allozymes that are more active at colder or warmer temperatures. Allozymes dominant in northern populations are more active in cold water.	Sockeye salmon	Kirpichnikov and Ivanova (1977)
Esterase-2 locus		Pink salmon	Kirpichnikov and Ivanova (1977)
Isocitrate dehydrogenase		Arctic char	Nyman and Shaw (1971)
Malic enzyme-2 locus		Steelhead trout	Redding and Schreck (1979)
Lactate dehydrogenase 5		Atlantic salmon	Verspoor and Jordan (1989)
		Brown trout	Henry and Ferguson (1985)
PHYSIOLOGICAL TRAITS			
Swim bladder function	Greater swim bladder gas retention ability in fish inhabiting deeper lakes.	Lake trout	Ihssen and Tait (1974)
Swimming ability	Superior prolonged swimming ability in stocks with long freshwater migrations.	Steelhead trout Coho salmon	Taylor and McPhail (1985)
Temperature tolerance	Resistance of fish naturalized to warm water environments and to high temperatures.	Rainbow trout	Morrissy (1973)
Time to smolting	More rapid development adapted to streams with short growing seasons.	Atlantic salmon (S)	Jensen and Johnsen (1986)
DISEASE RESISTANCE			
Resistance to *Cerafornyxa shasta*	Populations that have coevolved with *C. Shasta* have greater resistance than those that have not.	Chinook salmon Coho salmon Steelhead trout	Zinn et al. (1977) Hemmingsen et al. (1986) Buchanan et al. (1983)

Table 4-4. Examples of local variation in traits of salmonids and their presumed adaptive advantages.*

Trait	Adaptive Advantage	Species (Life cycle phase)†	Source
LIFE HISTORY TRAITS			
Large size	Larger size of adults adaptive in populations undertaking difficult migrations or experiencing high flows during spawning.	Chinook salmon Sockeye salmon Chum salmon Brown trout Atlantic salmon Pink salmon	Taylor (1990a) Rogers (1987) Beacham and Murray (1987) L'Abee-Lund et al. (1989) Power (1986) Rogers (1987); Beacham and Murray (1988)
Small size	Adaptation to streams with low summer flows.	Brown trout (J)	Borgstrom and Heggenes (1988); Titus and Mosegaard (1989)
Precocious males/parr maturation	Increased incidence of precocious males or parr maturation may be adaptive in populations with long, difficult migrations.	Chinook salmon Atlantic salmon	Myers et al. (1986) Taylor (1989)
Egg size	Late spawners tend to produce smaller faster-developing eggs than early spawners: facilitates synchronous emergence.	Chum salmon	Beacham and Murray (1987) Fleming and Gross (1990)
Fecundity	High fecundity favored in populations that experience high pre-reproductive mortality.	Chinook salmon (A)	Healey and Heard (1984)

* Examples are from a review by Taylor (1991).

† Life cycle phases: E = embryo/alevin, F = fry, J =juvenile, S = smolt. A = adult

taxonomic groups suggests that the probability of a local extinction increases with decreasing population size, decreasing size of habitat patches, and increasing isolation from other conspecific populations (reviewed in Hanski 1991; Sjogren 1991). The risk of extinction is also believed to be greater for populations that undergo large natural fluctuations in abundance (Harrison 1991). Recolonization rates are similarly influenced by population size and distance between habitat patches. Re-establishment of populations depends on sufficient numbers of individuals invading that habitat, which in turn depends on dispersal rates, the population size of source populations, the proximity and size of nearby habitat patches, and the availability of suitable migration corridors between patches.

Salmonid metapopulations exhibit many characteristics that would appear to make them vulnerable to extinction. Nehlsen et al. (1991) identified 101 stocks of anadromous salmonids that have had escapements under 200 within the last 1 to 5 years. These stocks are at increased risk of extinction from stochastic genetic, demographic, or environmental events. Many extant salmonid stocks have been eliminated from lower-elevation stream reaches and persist only as remnant populations confined to smaller headwater streams that have been less affected by habitat alterations. First- and second-order streams in steep headwaters tend to be hydrologically and geomorphically more unstable than larger, low-gradient streams. Thus, salmonids are being restricted to habitats where the likelihood of extinction because of random environmental events is greatest. If salmonid metapopulation structure historically resembled the core-satellite model, important source populations may already have been lost, leaving primarily extinction-prone satellite populations, Increased fragmentation of aquatic habitats and isolation of salmon populations reduces the chances that straying individuals from other populations can help restore depleted stocks. Snake River sockeye salmon provide a good example of an isolated population that is unlikely to be rescued by strays from other populations, since the nearest sockeye stock is over 900 km away (Waples 1995) and straying rates are low. Lastly, salmonids have historically experienced wide interannual variation in numbers as a consequence of variation in both freshwater and marine conditions. Numbers of coho salmon returning to streams in Oregon, Washington, and California can vary by an order of magnitude or more in different years (Hall and Knight 1981). Similar variability in escapement of pink and sockeye salmon has also been documented (Burgner 1991; Heard 1991). The probability of extinction because of fluctuating numbers combined with random environmental events may be particularly high for those species such as pink and coho salmon that have comparatively rigid life histories. In these species, the loss of a particular year-class may have longer-lasting effects than in populations with greater diversity in the age of spawning adults.

A final aspect of metapopulation theory that is relevant to salmonid conservation relates to temporal difference in the dynamics of the local populations that constitute the metapopulation. Hanski (1991) proposed that metapopulation persistence should be greatest where local populations fluctuate independently of each other, i.e., asynchronously, and lowest where local populations fluctuate synchronously in response to regional environmental conditions. The widespread declines in salmon populations throughout the Pacific Northwest suggest that fluctuations in these populations are synchronous, therefore, the risk of metapopulation extinction is relatively high.

4.2.5 Evolutionarily Significant Units

Under the Endangered Species Act or ESA (as amended in 1978), a "species" is defined to include "any subspecies of fish or wildlife or plants, or any distinct population segment of any species of vertebrate fish or wildlife which interbreeds when mature." For anadromous Pacific salmon and trout, most stocks are, to varying degrees, reproductively isolated—and hence potentially distinct population segments—but ESA provides no direction for determining what constitutes a distinct population segment (Waples 1995). To address this concern, the National Marine Fisheries Service (NMFS) has adopted a definition of "species" that is based on the concept of "evolutionarily significant units" or "ESUs" (Waples 1991b). A population is considered an ESU if it meets two criteria: 1) it is substantially reproductively isolated from other conspecific population units, and 2) it represents an important component in the evolutionary legacy of the species (Waples 1991b). For the first criterion to be met, isolation of the population need not be complete, but it must be sufficient to allow accrual of differences in specific traits among populations. Nor is isolation by itself sufficient for a population to be considered distinct. A population may meet the second criterion if it contributes to the overall genetic diversity of the species. In addition, because ecological diversity may foster local adaptations, stocks occupying distinct or unusual habitats or that are otherwise ecologically distinct may also be ESUs (Waples 1991b).

The intent of the ESU framework is to conserve the genetic diversity of species and the ecosystems that species inhabit, two fundamental goals of ESA (Waples 1991b). The genetic variability within a stock or population represents both the legacy of past evolutionary events and the ability of the population to respond to future environmental changes. The loss of individual stocks or the alteration of the genetic composition of stocks through hatchery introductions can fundamentally alter the ability of the species to cope with local environmental conditions, to respond to environmental change, and hence to persist over the long term.

Waples (1991b) advocates a two-step approach for determining whether a population represents a distinct unit. The first step is to evaluate the degree of reproductive isolation of the population. With salmonids, and particularly anadromous forms, reproductive isolation is rarely complete because of straying and is more a matter of degree. Waples (1991b) recommends several approaches for assessing the degree of reproductive isolation including 1) use of tags to estimate straying rates, 2) intentional genetic marking of populations, 3) use of genetic indices to estimate levels of gene flow, 4) observation of recolonization rates, and 5) identification of physical or geographic features likely to act as barriers to migration. The second step is to evaluate whether the population exhibits evidence of substantial ecological or genetic diversity. Factors to consider include 1) genetic traits, including unique alleles, different allelic frequencies, total genetic diversity; 2) phenotypic traits, including morphological or meristic characters, occurrence of parasites, and disease or parasite resistance; 3) life-history traits, such as time, age, or size at spawning, fecundity, migration patterns, and timing of emergence and outmigration; and 4) habitat characteristics, including temperature, rainfall, streamflow, water chemistry, or biological attributes of the particular system (Waples 1991b).

As Waples (1991b) notes, interpreting data for reproductive isolation is not always straightforward. For example, assessments of straying rates may be confounded by behavior of migratory adults (e.g., temporary entry of fish into non-natal streams). Measures of gene flow may require assumptions of selective neutrality for the alleles used. Assessment of allelic frequencies or presence of unique alleles may be influenced by sampling design, including number of samples and time of sampling. Similarly, interpreting ecological or genetic diversity data may be equally difficult. Variation in phenotypic and life-history characteristics may be attributable to both genetic and environmental factors; thus phenotypic or life-history variation alone is insufficient for determining population distinctness. The ability to distinguish distinct and unusual habitats is limited by both science and differences of opinion as to what are important habitat characteristics.

Identification of evolutionarily important biological units for protection is further complicated by the fact that a significant number of salmonid stocks have already been lost, and as a result, our understanding of metapopulation structure and function is incomplete. Li et al. (1995) note that few high-quality habitats remain and that many of these lie at the extremes of species' ranges. They argue that conservation strategies should differ depending on metapopulation structure. For example, the classical metapopulation model (Levins 1969) assumes that populations within each metapopulation each carry equal "evolutionary weight," whereas the "core-satellite" model proposes that "core" populations are critical for maintaining smaller satellite populations that might otherwise go extinct. With the classical metapopulation model, the best conservation strategy might be to treat all populations as equally important, protecting as many unique populations as possible in order to protect diversity. In the core-satellite system, emphasis should be placed on protecting core populations, since failure to do so would result in marginal populations of narrow specialists occupying the extremes of the species' range (Li et al. 1995). Waples (1991b) similarly argues that threatened and endangered status should be considered for metapopulations as well as more discrete population units.

Finally, an assumption of the ESU concept is that not all populations need to be protected in order to preserve the genetic integrity of the species (Waples 1991b). Local populations that are not reproductively isolated or that are isolated but fail to exhibit any important and distinctive genetic or life-history traits do not qualify for protection under ESA. In practice, such populations are typically not genetically differentiable from hatchery populations. Where such populations are lost, their ecological function in the aquatic community will also be lost and other organisms may be affected over the evolutionary short term. However, over longer evolutionary time scales the ESU conservation strategy will result in available habitats repopulated by native fish from either within the local ESU or from neighboring ESUs. This should result in fish populations locally adapted or more able to survive and reproduce in the wild, thereby fulfilling their role in the ecosystem.

4.3 Community Level

The biotic communities of aquatic systems are highly complex entities. Within communities, assemblages and species have varying levels of interaction with one another. Direct interactions may occur in the form of predator-prey, competitor, and disease- or parasite-host relationships. In addition, many indirect interactions may also occur between species. For example, predation of one species upon another may enhance the ability of a third species to persist in the community by releasing it from predatory or competitive constraints. These interactions continually change in response to shifting environmental and biotic conditions. Human activities that modify either the environment, the frequency and intensity of disturbance, or species composition can shift the competitive balance between species, alter predatory interactions, and change disease susceptibility, all of which may result in community reorganization.

The role of disturbance in regulating stream-community organization has been a principal focus of aquatic ecology in the past decade. In a recent review, Resh et al. (1988) identify three theories (equilibrium, intermediate disturbance hypothesis, dynamic equilibrium model) that reflect our present understanding of disturbance theory as it relates to stream-community structure. The equilibrium theory proposes that environments are more or less constant and that community organization is determined by biotic interactions, including competition, mutualism, and trophic interactions. The intermediate disturbance hypothesis assumes a competitive hierarchy of species. In the absence of disturbance, superior competitors eliminate inferior ones, whereas in systems with frequent or severe disturbances, resident competitors are eliminated and colonizing species dominate. In systems with intermediate disturbance regimes, species richness is maximized; colonizers exploit disturbed areas and are thus able to coexist with superior competitors. The dynamic equilibrium model proposes that community structure is a function of growth rates, rates of competitive exclusion, and frequency of population reductions. Inferior competitors persist in the community if disturbances occur often enough to eliminate competitive exclusion; however, if disturbances are too frequent, species with long life cycles are eliminated. Species diversity is determined by the influence of the environment on the net outcome of species interactions. Both the intermediate disturbance hypothesis and dynamic equilibrium model emphasize the role of the environment in regulating stream communities, and Resh et al. (1988) conclude that these hypotheses are more generally applicable to stream ecosystems than the equilibrium model. All three models may be applicable depending on spatial and temporal scales and the type of aquatic system (streams, lakes, reservoirs, and estuaries).

4.3.1 Food Webs

The food energy available to fish and other organisms in aquatic ecosystems comes from two sources: aquatic plants (macrophytes, benthic algae, and phytoplankton) that convert solar energy into biomass and organic litter that falls into the stream and provides the energy base for fungi and bacteria (OWRRI 1995). The relative importance of these energy sources changes with the size and morphology of a river, estuary, or lake system (see Section 4.4.2) and the availability of nutrients in the catchment. Herbivorous aquatic invertebrates consume algae and other aquatic plants, whereas detritivorous invertebrates consume decaying organic matter. Many invertebrates select food on the basis of size, rather than source, while others are generalized feeders. Predatory invertebrates may add an additional trophic level to the food web. Collectively, these invertebrates form an important food base for many juvenile anadromous salmonids and adult trout although some species may feed on other fishes and terrestrial insects that fall into the stream. Fishes, in turn, are consumed by a host of terrestrial and aquatic predators, including other fishes, birds, mammals, reptiles, and amphibians. The resulting food webs can be highly complex, consisting of many species representing several trophic levels.

Food webs may be highly modified by environmental changes, including alterations of the food base; changes in streamflow, temperature, and substrate; and the introduction of non-native organisms. Alterations of individual components of a food web can propagate throughout the system, leading to community wide adjustments in food web composition. For example, impoundments on the Columbia River have shifted the food base from coarse detrital material derived from wetland emergent vegetation and fine material derived from periphyton to a phytoplankton-derived micro-detritus food base, creating numerous adjustments throughout food web (Simenstad et al. 1990; Palmisano et al. 1993b). In the estuary, amphipods and isopods—the preferred food items of salmonid smolts (Dawley 1986)—have now been replaced by suspension feeding epibenthos (Simenstad et al. 1990), which are a primary food source for juvenile American shad. An increasingly favorable environment for shad, coupled with relatively low predation rates, **has** allowed the population to increase dramatically over the last few decades from less than 200,000 to approximately 4 million (Palmisano et al. 1993b). Elimination of woody riparian vegetation from rangeland streams has shifted the food base from coarse, terrestrially derived material to periphyton. The latter is most efficiently consumed by shell-cased macroinvertebrates that are unsuitable prey for juvenile salmonids (Tait et al. 1994).

Changes in water temperatures may change the composition of algal assemblages (Bush et al. 1974); disrupt the development and life-history patterns of benthic macroinvertebrates (Nebeker 1971; Lehmkuhl 1972) and zooplankton (Hutchinson 1967); and decrease the abundance of certain benthic invertebrates, especially species that are stenotherms (Hynes 1970).

Introductions of non-native fish, either as game fish or forage for other fish, have led to food web alterations in most river systems of the Pacific Northwest. In California and Oregon, introduced fishes constitute 35% and 29% of the total species, respectively. The impact of these fish on native species is poorly known, but they are potential predators and competitors of both the juvenile and adult salmonids. The mainstem Columbia river is host to numerous non-native fish, many of them piscivorous, that have acclimated to the lentic habitat of the reservoirs and now dominate many of the trophic pathways. Several mechanisms have been identified that allow introduced fish to succeed in displacing native species, including competition, predation, inhibition of reproduction, environmental modification, transfer of new parasites or diseases, and hybridization (Moyle et al. 1986). Similarly, introductions of invertebrates can modify food webs. The introduction of opossum shrimp to Flathead Lake, Montana, resulted in the disappearance of two cladoceran species, which in turn had negative effects on the kokanee salmon that were intended to benefit from the introduction (Spencer et al. 1991).

4.3.2 Competition

Competition among organisms occurs when two or more organismic units (i.e., individuals or species) use the same resources and when availability of those resources is limited (Pianka 1978). Two types of competition are generally recognized: interference competition, where one organism directly prevents another from using a resource through aggressive behavior, and exploitation competition, where one species affects another by utilizing a resource more efficiently (Moyle et al. 1986). Although competition is difficult to demonstrate (Fausch 1988), salmonids likely compete for food and space resources both within species (intraspecific) and between species (interspecific). Within species, stream-dwelling salmonids frequently form dominance hierarchies, with dominant individuals defending holding positions against subordinate fish through agonistic encounters. Evidence suggests that dominant individuals occupy the most energetically profitable holding positions, which in turn leads to higher growth rates (Fausch 1988). Similar interactions occur between salmonid species; however, in assemblages that have co-evolved, species with similar ecological requirements may segregate according to both micro- and macrohabitats at various life stages.

Changes in physical (e.g., temperature, streamflow, habitat structure) and biological (e.g., food availability, species composition) characteristics of streams and lakes can alter competitive interactions within and among species, potentially resulting in a restructuring of fish communities. In a laboratory study, Reeves et al. (1987) found that stream temperature affected interspecific interactions between juvenile steelhead trout and redside shiner *(Richardsonius* balleatus)—with trout competing more effectively at cold temperatures through interference and shiner competing more successfully at warm temperatures through both exploitation and interference. Cunjak and Green (1986) found that interactions between brook trout *(Salvelinus fontinalis)* and rainbow trout are also influenced by water temperature, rainbow trout being superior competitors at 16°C and brook trout at 9°C. Ratliff (1992) suggests that the decline of bull trout populations in Oregon may in part reflect the inferior ability of bull trout to compete with rainbow, brook, and brown trout *(Salmo trutta)* at warmer temperatures.

Changes in streamflow in the Columbia River system have resulted in increased plankton production, which has apparently increased the success of American shad. Palmisano et al. (1993a, 1993b) conclude that increased numbers of shad, which also feed on benthic invertebrates, may be competing with juvenile salmonids. Cunjak and Green (1984) reported that brook trout tended to dominate social interactions with rainbow trout when in pool habitats, but not in faster waters. Larson et al. (1995) suggest that the dynamics of brook trout and rainbow trout interactions in a southern Appalachian stream may be affected by both temperature and flow conditions. During years of low discharge, rainbow trout encroached on upstream habitats of brook trout possibly because warmer temperatures favored rainbow trout. During periods of higher discharge, encroachment was reversed, presumably because brook trout are better adapted to the steep stair-stepped channel morphology. In general, decreases in streamflow decrease available habitat and may thereby intensify inter- and intraspecific competition for suitable rearing, feeding, spawning, and refuge habitats.

The introduction of non-native species increases the potential for competition in aquatic systems. In natural fish assemblages, salmonids have presumably adapted to other native species with similar ecological requirements through resource partitioning or segregation in time or space. With the introduction of non-native species, however, there has been no opportunity for natural selection to ameliorate competition (Fausch 1988). Several studies have documented influences of non-native species on native salmonids. In a British Columbia lake, cutthroat trout were found to shift from midwater areas when allopatric to littoral zones when sympatric with rainbow trout (Nilsson and Northcote 1981). Dambacher et al. (1992) found that non-native brook trout outcompeted bull trout in Sun Creek, Oregon, in areas of co-occurrence. Intraspecific interactions may also become more intense with the introduction of hatchery fish. Nickelson et al. (1986) concluded that competition between larger hatchery coho salmon and wild juveniles resulted in 44% replacement of the wild fish.

4.3.3 ***Predation***

Adult and juvenile salmonids have evolved strategies to coexist with numerous natural predators including a variety of fish, birds, and mammals. Native fish piscivores include sculpin *(Cottus* spp.), bull trout, rainbow trout, cutthroat trout, northern squawfish *(Ptychochilus oregonensis)*, and possibly white sturgeon *(Acipenser transmountanus)*. These fish prey on juvenile salmonids during instream rearing and during smolt migrations. Northern squawfish are considered important predators of outmigrant salmon and steelhead trout particularly in slackwater habitats (Poe et al. 1991). Bird predators of smolts and juveniles (Kaczynski and Palmisano 1993) include ring-billed gulls *(Larus delawarensis)*, common mergansers *(Mergus merganser)*, herons *(Ardea* spp.), and kingfishers *(Megaceryle alcyon)*. Kingfishers were found to have increased feeding efficiency in slower moving waters. Pinnipeds, including harbor seals *(Phoca vitulina)*, California sea lions *(Zalophus californianus)*, and Stellar sea lions *(Eumetopiajubatus)* are the primary marine mammals preying on salmonids, although Pacific striped dolphin *(Lagenorhynchus obliquidens)* and killer whale *(Orcinus orca)* may also prey on adult salmonids. Seal and sea lion predation is primarily in saltwater and estuarine environments though they are known to travel well into the freshwater environment after migrating fish. All of these predators are opportunists, searching out locations where juveniles and adults are most vulnerable.

Habitat alterations can affect predation rates by reducing cover, which increases vulnerability to capture by predators; altering flow regime and water velocity, which may favor certain piscivorous fishes; modifying temperature, which affects the metabolism of piscivorous fish and the ability of fish to elude predators; and by obstructing passage, which may delay migrations and thereby increase exposure to predators. In the Columbia Basin altered flow regimes have contributed to the increased success of northern squawfish, walleye *(Stizostedion vitreum),* and smallmouth bass (*Micropterus dolomieu*), which tend to avoid areas of high water velocity (Faler et al. 1988). Laboratory experiments with squawfish have shown that maximum consumption of salmonid smolts increased from 0.5 smolts per day at 8.3°C to 7 smolts per day at 21.7°C (Vigg and Burley 1991), indicating that temperature increases may indirectly cause greater predation on juvenile salmonids (Palmisano et al. 1993b). The high incidence of predation by sea lions at such places as Ballard Locks in Washington is in part attributable to the unnatural congregations of fish as they attempt to pass through the locks.

4.3.4 Disease and Parasitism

Salmonids are affected by a variety of bacterial, viral, fungal, and microparasitic pathogens. In the Pacific Northwest, numerous diseases may result from pathogens that occur naturally in the wild or that may be transmitted to wild fish via infected hatchery fish. Among these are bacterial diseases, including bacterial kidney disease (BKD), columnaris, furunculosis, redmouth disease, and coldwater disease; virally induced diseases, including infectious hepatopoietic necrosis (IHN) and erythrocytic inclusion body syndrome (EIBS); protozoan-caused diseases, including ceratomyxosis and dermocystidium; and fungal infections, such as saprolegnia (Bevan et al. 1994a). Brief descriptions of the most prevalent pathogens and the associated diseases are shown in Table 4-5.

Water temperature greatly influences the immune system of fishes, the number and virulence of pathogens, and in the case of microparasites, the occurrence of infective life stages in natural and aquacultural environments. Consequently, changes in water temperatures caused by forest and range practices, dams, and irrigation can alter the susceptibility of salmonids to infection by these pathogens. Most work on fish pathogens has concerned fish in culture situations, and the incidence of disease and its role of fish population dynamics and in structuring fish assemblages in natural waters is poorly understood (Austin and Austin 1993). Nevertheless, laboratory studies indicate that water temperature has a direct effect on the infection rate of most pathogens and the mortality rate of infected salmonids. With most pathogens, the susceptibility of salmonids to infection tends to increase with increasing water temperatures, although mortality from coldwater disease is greater when temperatures are lower (Holt et al. 1993). A summary of the general relationship between temperature and important pathogens in Pacific Northwest environments is shown in Table 4-5.

Several recent epizootics indicate that temperature may play a significant role in mediating disease in natural populations. Prespawning mortality in fall chinook salmon was highly correlated with mean maximum stream temperatures in the Rogue River (Oregon) during August and September, with mortality rates increasing abruptly at temperatures greater than 20°C (ODFW 1992). *Flexibacter colurnnaris* was commonly found in dead and dying fish and was presumed to be the primary agent causing mortality. Release of warm reservoir water during the late summer and early fall has been implicated in outbreaks of *Dermocystidium salmonis* in anadromous fish in the lower Elwha River, Washington (NPS et al. 1994). In 1992, approximately two-thirds of the adult chinook population in the lower river died prior to spawning (Wunderlich et al. 1994).

While epizootics provide the most dramatic examples of the potential for pathogens to affect salmonid populations, sublethal chronic infections can impair the ability of fish to perform in the wild and thereby contribute secondarily to mortality or reduced reproductive success. Fish weakened by disease are more sensitive to other environmental stresses. Furthermore, infected fish may become more vulnerable to predation (Hoffman and Bauer 1971), or less able to compete with other species. For example, Reeves et al. (1987) found that the interspecific interactions between juvenile steelhead trout and redside shiner were affected by water temperature. They speculated that these differences were in part because most juvenile steelhead were infected with *F. columnaris* at high temperatures, whereas shiners showed a higher incidence of infection at lower temperatures.

The susceptibility of salmonids to disease may be affected by other stressors, including dissolved oxygen, chemical pollution, and population density. Temperature may interact synergistically with these factors, causing disease to appear in organisms that might be resistant in the absence of other forms of stress. Susceptibility also varies among salmonid species and life stages. For example, older chinook have been shown to be more resistant to ***F.*** *colurnnaris* than younger fish (Becker and Fujihara 1978).

4.4 Connectivity Among Processes

The biotic communities found in streams and rivers reflect physical and chemical gradients that occur both across the landscape and along a stream from the headwaters to the ocean. In the preceding sections, we have reviewed fundamental biological processes that occur at the level of organisms,

Table 4-5. Pathogens of salmonids found in Pacific Northwest waters.

Pathogen	Disease	Comments
BACTERIA		
Aeromonas salmonicida	Furunculosis	Obligate pathogen of fish. Low mortality at temperatures < 6.7°C. Increasing mortality at 9.4°C. At 20.5°C, 93%–100% mortality for all species. (Groberg et al. 1978)
Aeromonas hydrophila		Stress facilitated infection. Mortality is associated with elevated water temperatures (> 9.4°C), presence of pollutants (particularly nitrate at ≥ 6 mg/L). (Austin and Austin 1993)
Flexibacter columnaris	Columnaris	Low mortality at temperatures < 15°C. Increasing mortality at 20°C for all species. Virulence at low temperatures depends on specific strain. Naturally occurring bacteria present at low levels in resident fish (suckers, carp, and whitefish). Stress increases fish susceptibility. High density increases potential for contact. (Inglis et al. 1993).
Flexibacfer psychrophilus	Coldwater disease	Appears in spring when temperatures are between 4–10°C; 30%–50% mortality for infected alevins; Quickest mortality at 15°C. Mean time to death increases with temperatures from 15–23°C. Mode of transmission unknown. Resident salmonids are probable carriers. Possible vertical transmission. (Inglis et al. 1993)
Renibacfenum salmoninarum	Bacterial kidney disease (BKD)	Obligate pathogen of fish. Disease progresses more rapidly at higher temperatures (15–20.5°C), but mortality may be highest at moderate temperatures (12°C). Transmission is both horizontal and vertical (intraovum). Crowding and diet stress can increase susceptibility. (Inglis et al. 1993; Fryer and Lannan 1993)
Yersinia rucken	Redmouth disease	Mortality may be low in chronic infections but becomes much higher with stress from poor water conditions (elevated temperatures, ammonia, metabolic waste, copper). Transmission through water, via baitfish, introduced fish, bird feces, fish farms. (Inglis et al. 1993)
FUNGI		
Saprolegnia		Ubiquitous in water. Transmitted horizontally or from substratum to fish. Elevated temperatures increase growth rate. If untreated, progressive and terminal. (Bell 1986)

Table 4-5. Pathogens of salmonids found in Pacific Northwest waters.

Pathogen	Disease	Comments
PROTOZOANS		
Ceratomyxa shasta	Ceratamyxosis	Endemic to many river systems of Northwest. Temperature dependent; increasing mortality for fish exposed at temperatures ≥ 10°C. High mortality for nonadapted (no genetic resistance) species and stocks. (Ratliff 1983).
Dermocystidium salmonis		Pathogen of salmonids in Pacific Northwest. Horizontal transmission through water. Uptake is via gills. Epizootics appear to be temperature dependent. (Olson et al. 1991)
Ichthyobodo/Costia spp.		Ectoparasite affects osmoregulation. Juvenile salmonid mortality high (63%–70% in 48 h tests) upon introduction to marine waters. (Urawa 1993)
Myxobolus cerebralis	Whirling disease	Salmonid infection by mature triactinomyxon via ingestion or through gills. Horizontal transmission. Intermediate host is tubifex worm from soft mud habitats. Lethal to salmonids. (Rich Holt, personal communication, 1995).
VIRUSES		
infectious Hepatopoietic Necrosis Virus	IHN	Endemic to most areas. High for young fry. Most mortality occurs at temperatures of 12°C or less. Some outbreaks at 15°C. At temperatures over 10°C, disease produces less mortality but leads to more carriers of disease. (Wolf 1988).
Erythrocytic Inclusion Body Syndrome Virus	EIBS	Potential vertical transmission and known horizontal transmission. Greatest mortality of salmonids found at 8–10°C. (Takahashi et al 1992; Leek 1987)

populations, and communities, and the relationship between these processes and habitat characteristics that are affected by human activities. In this section, we discuss two concepts, the river continuum concept and the ecoregion concept, that address spatial relationships between these physicochemical and biological processes. The river continuum concept (Vannote et al. 1980) focuses on interrelationships between physical and biological processes along streams from their headwaters to the ocean. The ecoregion concept relates regional patterns in physical and chemical gradients to the biological communities contained therein.

4.4.1 River Continuum Concept

The river continuum concept (Vannote et al. 1980) proposes that the physicochemical variables (e.g., light, nutrients, organic materials) within a river system change in a systematic way as a stream flows from headwaters to larger river systems to the ocean, and that the biological communities found along this gradient change accordingly (Figure 4-2). In forested headwater reaches, energy inputs are dominated by coarse allochthonous materials, particularly leaf litter from riparian vegetation. As streams increase in size, canopy cover becomes less complete and more light reaches the stream; consequently, the contribution of instream primary

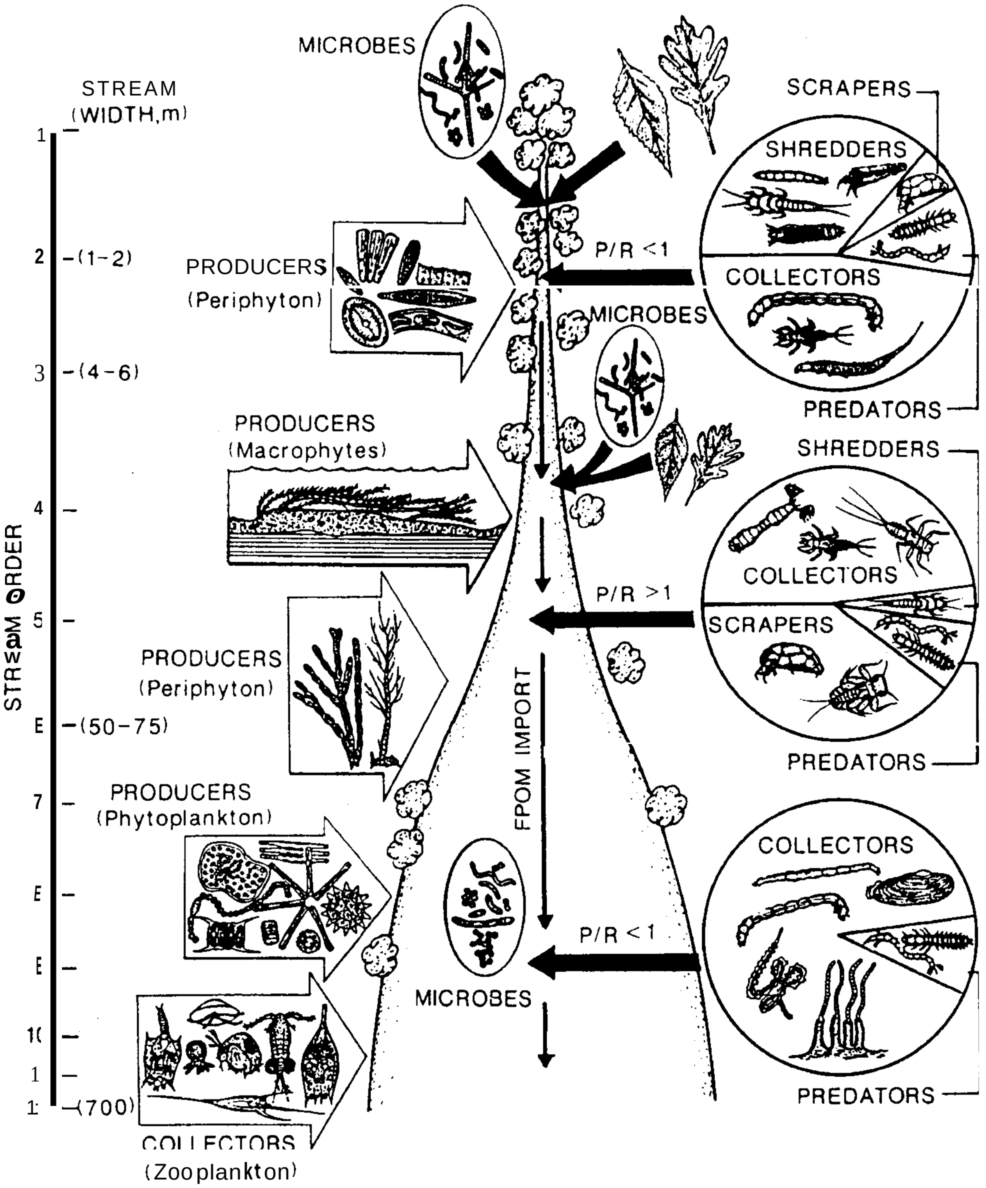

MICROBES
STREAM (WIDTH,m)
1
2 -(1-2)
3 -(4-6)
4
5
6 -(50-75)
7
8
9
10
11
12 -(700)
STREAM ORDER
PRODUCERS (Periphyton)
PRODUCERS (Macrophytes)
PRODUCERS (Periphyton)
PRODUCERS (Phytoplankton)
COLLECTORS (Zooplankton)
P/R <1
MICROBES
P/R >1
FPOM IMPORT
P/R <1
MICROBES
SCRAPERS
SHREDDERS
COLLECTORS
PREDATORS
SHREDDERS
COLLECTORS
SCRAPERS
PREDATORS
COLLECTORS
PREDATORS

Figure 4-2. Trends in energy sources, ratios of autotrophic production to heterotrophic respiration, and functional groups along a river continuum. From Vannote et al. (1980). Reproduced with permission from the publisher.

production from algae and macrophytes increases relative to energy derived from allochthonous materials. In still larger systems, fine particulate material transported from upstream areas forms the dominant energy source, particularly where depth and turbidity limit algal growth. These gradients lead to corresponding changes in the biological communities that use these changing energy sources. Invertebrate communities shift from those dominated by shredders and collectors in small streams, to collectors and grazers in mid-order streams, to mostly collectors in large rivers. Fish assemblages shift from invertivores in headwater reaches, to piscivores and invertivores in mid-order reaches, and include some planktivores in larger rivers.

Although the river continuum concept was developed in forested biomes, it can also be applied more generally. Meehan (1991) suggests that meadows and deserts, which lack shading and have reduced allochthonous inputs, obtain most of their energy from autochthonous sources, in contrast to woodland streams which have stronger terrestrial influences and therefore greater quantities of coarse particulate detritus. They conclude that desert streams are more similar to the downstream reaches of forested streams. Minshall et al. (1985) illustrate this conceptually by proposing a sliding scale to indicate that streams enter the continuum at different points. Similarly, primary production by algae may be high in headwater streams of alpine systems, where riparian inputs are comparatively low. Consequently these systems may have a different sequence in the biological communities along the continuum.

4.4.2 Ecoregions

Ecological processes that influence salmonids and other aquatic species in the Pacific Northwest vary greatly across the landscape because of the high diversity of climate, topography, geology, vegetation, and soils. Classifications of ecoregions represent attempts to identify areas of relative homogeneity in ecological systems or in the relationships between organisms and their environments (Omernik and Gallant 1986). Several Federal agencies, including the Environmental Protection Agency (Omernik and Gallant 1986; Omernik 1987), the U.S. Forest Service (Bailey 1978), and the Soil Conservation Service (Norris et al. 1991) have developed or are in the process of developing ecoregion classifications in order to address spatial issues in the management of natural resources. Landscapes, water bodies, and the biota that they support are expected to be similar within an ecoregion and to differ between ecoregions. We believe some form of ecoregion classification will be essential to defining the natural range of physical, chemical, and biological characteristics of salmonid habitats across the landscape.

The various processes for delineating ecoregions differ. Omernik and his colleagues synthesized a number of factors (climate, geology, topography, soil, vegetation, land cover) to assess patterns at multiple spatial scales. Bailey considered many of the same factors but used only one at any single scale of resolution. For example, his first divisions were by climatic patterns and his last were by vegetation. The Soil Conservation Service, as might be expected, focused on soil and agricultural land uses. Omernik's approach is favored by many State water quality agencies because of its ability to assess patterns at multiple scales and its adaptability, and it has been recommended by other scientific organizations (SAB 1991; NRC 1992).

Although there are serious limitations to the application of Omernik's ecoregions at the site or small catchment scales, they are useful for stratifying the regional variability of the Pacific Northwest (Table 4-6) into relatively distinct units. In addition, ecoregions offer a framework for aggregating and extrapolating data collected at the local level. A regional perspective is also essential for managing widely distributed resources, such as Pacific salmonids, because of the natural variability among sites and the human tendency to focus on local issues while losing sight of regional ones. In addition, subregions can be developed in a hierarchical manner to facilitate more precise landscape classification at local scales (Clarke et al. 1991; Bryce and Clarke 1996). Direct applications of ecoregion concepts to aquatic ecosystems have demonstrated the utility of this approach. Whittier et al. (1988) showed that fish assemblages in rivers and small streams exhibited patterns concordant with Omernik's ecoregions in Oregon. In evaluating a number of different data sets from basin to State scales, Hughes et al. (1994) found ecoregions that differed markedly supported dissimilar fish assemblages, similar ecoregions supported more similar fish assemblages, and within-region variation was less than among-region variation.

4.5 Summary

In the preceding sections, we have discussed biological processes at three levels of biological organization: organisms, populations, and communities. Grouping processes into these discrete categories serves to simplify thinking about the effects of environmental perturbations on salmonids and their ecosystems, but it should be reiterated that salmonids are simultaneously affected by processes occurring at all levels of biological organization. Physiological stresses influence the ability of salmonids to acquire food and defend space from competitors, to escape or avoid predators, and to fend off infectious diseases and parasites, all of

Table **4-6.** Predominant characteristics of ecoregions in the Pacific Northwest. From Omernik and Gallant (1986).

Ecoregion	Land surface form	Potential natural vegetation	Land use	Soils*
Coast Range	Low to high mountains	Spruce/cedar/hemlock, cedar/hemlock/Douglas-fir. redwood	Forest and woodland mostly ungrazed	Udic soils of high rainfall areas
Puget Lowland	Tablelands with moderate relief, plains with hills or mountains, or open hills	Cedar/hemlock/Douglas-fir	Mosaic including forest, woodland, pasture, cropland	Alfisols, Inceptisols, Mollisols, Spodosols, and Vertisols of valleys
Willamette Valley	Plains with hills, or open hills	Cedar/hemlock/Douglas-fir, mosaic of Oregon oakwoods and cedar/hemlock/Douglas-fir	Primarily cropland with some interspersion of pasture, woodland, and forest	Xeric Mollisols, Vertisols, and Alfisols of interior valleys
Cascades	High mountains	Silver fir/Douglas-fir, fir/hemlock, western spruce/fir, Douglas-fir, cedar/hemlock/Douglas-fir, spruce/cedar/hemlock	Forest and woodland mostly ungrazed	Udic soils of high rainfall mountains
Sierra Nevada	High mountains	Mixed conifer forest (fir, pine, Douglas-fir), red fir, lodgepole pine/subalpine forest (hemlock)	Forest and woodland grazed	Xeric soils of moderate rainfall areas
Southern and Central California Plains and Hills	Irregular plains, tablelands of moderate to considerable relief, low mountains	California oakwoods, chaparral (manzanita, ceanothus), California steppe (needlegrass)	Open woodland grazed	Light-colored soils of subhumid regions
Central California Valley	Flat plains	California steppe (needlegrass), tule marshes (bulrush, cattails)	Irrigated agriculture, cropland with grazing land	Recent alluvial soils, light-colored soils of the wet and dry subhumid regions
Eastern Cascades Slopes and Foothills	Varied: tablelands with moderate to high relief, plains with low mountains, open low mountains, high mountains	Western ponderosa pine	Forest and woodland grazed	Xeric soils of moderate rainfall areas
Northern Rockies	High mountains	Cedar/hemlock/pine, western spruce/fir, grand fir/Douglas-fir, Douglas-fir	Forest and woodland mostly ungrazed	Eastern interior mountain soils with acidic rock types, Inceptisols
Columbia Basin	Varied: irregular plains, tablelands with moderate to high relief, open hills (excludes extremes)	Wheatgrass/bluegrass, fescue/wheatgrass, sagebrush steppe (sagebrush, wheatgrass)	Mostly cropland, cropland with grazing land	Xerolls, channeled scablands
Blue Mountains	Low to high open mountains	Grand fir/Douglas-fir, western ponderosa pine, western spruce/fir, Douglas-fir	Forest and woodland grazed	Soils of eastern interior mountains, Mollisols, Inceptisols
Snake River Basin/High Desert	Tablelands with moderate to high relief, plains with hills or low mountains	Sagebrush steppe (sagebrush, wheatgrass), saltbushlgreasewood	Desert shrubland grazed, some irrigated agriculture	Aridisols, aridic Mollisols

* Soils are presented in this table as they appear from mapped units of resource soil maps.

which affect community structure. Populations have evolved specific mechanisms for coping with environmental conditions in their natal and rearing streams. These adaptations include morphological, biochemical, physiological, behavioral, and developmental traits that allow fish to survive and thrive with the specific physical, chemical, or biological constraints imposed by the environment and that ensure specific activities (e.g., timing of migration and emergence) coincide with favorable environmental and ecological conditions. Adaptation is also evident in life-history strategies (e.g., fecundity and straying rates) that accommodate natural disturbance regimes and allow populations to persist over evolutionary time. Unlike the biological diversity of fishes in the Mississippi Basin, which centers on species diversity, the fish diversity in the Pacific Northwest centers on stock and life-history diversity. The evolution of a wide variety of life-history strategies has allowed salmonids to invade and thrive in the diverse habitats of the Pacific Northwest. The linkage between biological communities and the physical and chemical characteristics of streams are illustrated through the concepts of the river continuum and ecoregions, which offer means for assessing patterns in aquatic community structure across the landscape and for predicting the response of aquatic ecosystems to anthropogenic disturbance. These concepts are essential in developing site-specific and region-specific salmonid conservation strategies and goals.

5 Habitat Requirements of Salmonids

Karr (1991) defines biological integrity as "the ability to support and maintain a balanced, integrated, adaptive community of organisms having a species composition, diversity, and functional organization comparable to that of natural habitat of the region." He further states that a biological system can be considered "ecologically healthy" when "its inherent potential is realized, its condition is stable, its capacity for self-repair when perturbed is preserved, and minimal external support for management is needed."

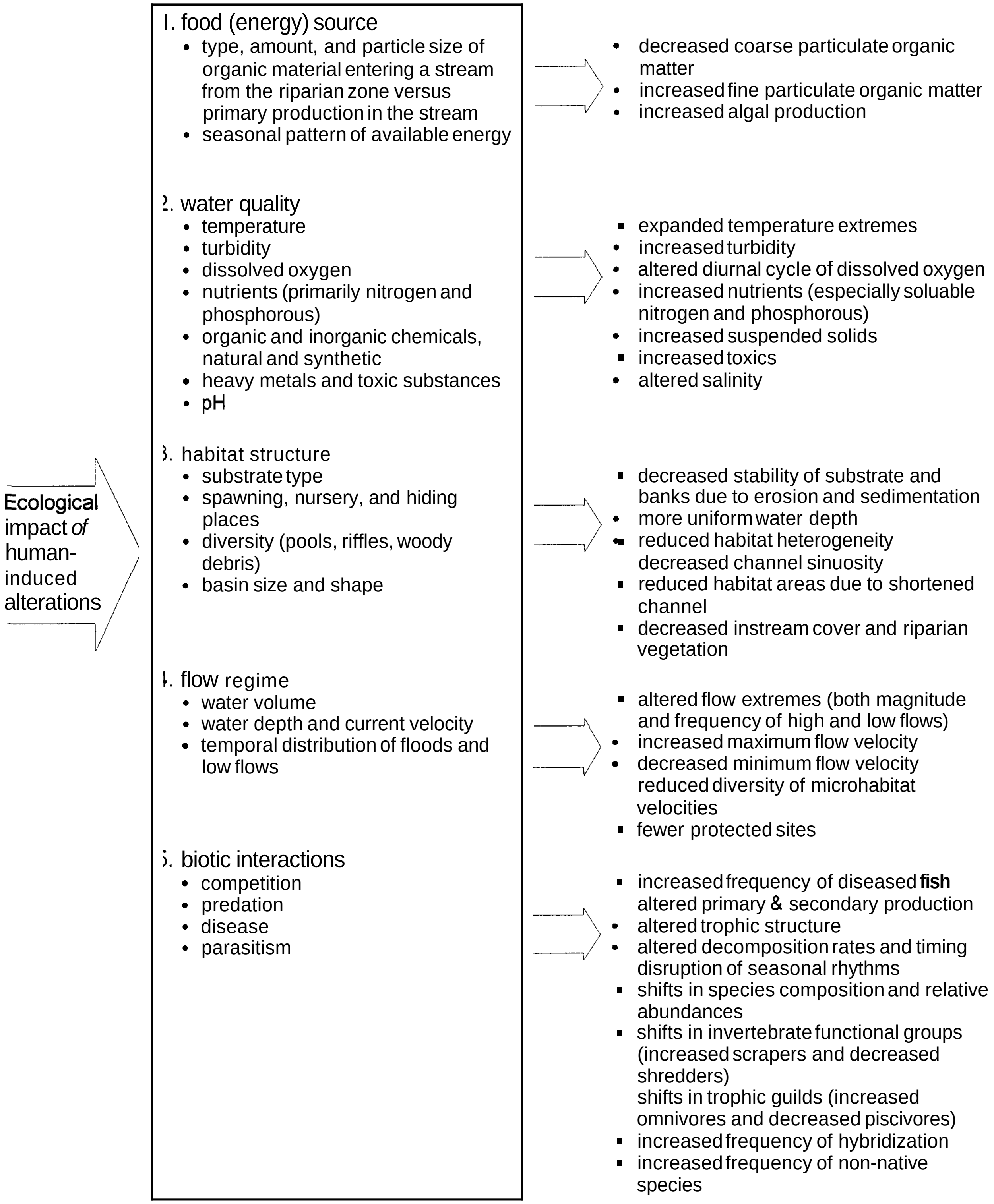
Specific attributes of streams and lakes, such as streamflow, water temperature, substrate, cover, and dissolved materials—all the elements typically associated with the term habitat—are the result of physical, chemical, and biological processes operating throughout a watershed and across the landscape (see Chapters 3 and 4). Protecting and restoring desirable habitat-attributes of streams and lakes for salmonids requires that the natural processes that produce these characteristics be maintained or restored. If processes are protected, in other words, desirable aquatic-habitat characteristics will develop; if the processes are altered, the integrity of the aquatic ecosystem and its ability to support salmonids are diminished. The Aquatic Conservation Strategy Objectives detailed by FEMAT (1993) directly reflects these concepts.

To assess the habitat requirements of salmonids, four principles need to be considered: 1) all watersheds and streams are different to some degree in terms of their temperature regimes, flow regimes, sedimentation rates, nutrient fluxes, physical structure, and biological components; 2) the fish populations that inhabit a particular body of water have adapted—biochemically, physiologically, morphologically, and behaviorally—to the natural environmental fluctuations that they experience and to the biota with which they share the stream, lake, or estuary (see Section 4.2.3); 3) the specific habitat requirements of salmonids differ among species and life-history types and change with season, life stage, and the presence of other biota: and 4) aquatic ecosystems are changing over evolutionary time. From these general principles, there are obviously no simple definitions of desirable habitat characteristics of salmonids. Defining acceptable or natural ranges of variability for specific habitat attributes is not only difficult, it can be misleading as well. For example, the same total sediment yield in two different watersheds may affect salmonid habitats differently, depending on geology, topography, hydrology, stream size, and the abundance of large woody debris. Similarly, Behnke (1992) has suggested that stocks of trout native to warmer streams may exhibit greater tolerance to high temperature extremes than stocks inhabiting naturally cooler waters; simply defining the range of temperatures at which a species has been observed does not ensure that stocks will be "safe" or healthy as long as temperatures remain in that range. The FEMAT (1993) report concluded that current scientific information is inadequate to allow definition of specific habitat requirements of salmonids throughout their life histories. These points further emphasize the need to maintain the integrity of aquatic ecosystems.

Karr (1991) identified five classes of environmental factors that affect the biotic integrity of aquatic ecosystems—food (energy) source, water quality, habitat structure, flow regimes, and biotic interactions—as well as ecological changes that may occur in response to human-induced alterations of these factors (Figure 5-1). Although this model was intended to address all aquatic biota, the elements provide a useful framework for discussing salmonid habitat requirements. In Section 5.1, we use the model of Karr (1991) to outline general habitat requirements of salmonids, focusing on processes and characteristics that must be maintained in order to ensure the ecological health of aquatic ecosystems. In Section 5.2, we review specific habitat requirements of Pacific salmonids at each life stage: adult migration, spawning and incubation, rearing, and juvenile migration. An extended discussion of water-quality concerns is presented under "general habitat requirements"; it is beyond the scope of this report to comprehensively review the effects of toxic substances on each life stage.

5.1 General Habitat Requirements

Everest et al. (1985) noted that although each species of anadromous salmonid differs somewhat in its specific habitat requirements, all share some common habitat needs. Extending their list to include resident species, all salmonids require sufficient invertebrate organisms for food; cool, flowing waters



1. food (energy) source
- type, amount, and particle size of organic material entering a stream from the riparian zone versus primary production in the stream
- seasonal pattern of available energy

- decreased coarse particulate organic matter
- increased fine particulate organic matter
- increased algal production

2. water quality
- temperature
- turbidity
- dissolved oxygen
- nutrients (primarily nitrogen and phosphorous)
- organic and inorganic chemicals, natural and synthetic
- heavy metals and toxic substances
- pH

- expanded temperature extremes
- increased turbidity
- altered diurnal cycle of dissolved oxygen
- increased nutrients (especially soluable nitrogen and phosphorous)
- increased suspended solids
- increased toxics
- altered salinity

3. habitat structure
- substrate type
- spawning, nursery, and hiding places
- diversity (pools, riffles, woody debris)
- basin size and shape

- decreased stability of substrate and banks due to erosion and sedimentation
- more uniform water depth
- reduced habitat heterogeneity
decreased channel sinuosity
- reduced habitat areas due to shortened channel
- decreased instream cover and riparian vegetation

4. flow regime
- water volume
- water depth and current velocity
- temporal distribution of floods and low flows

- altered flow extremes (both magnitude and frequency of high and low flows)
- increased maximum flow velocity
- decreased minimum flow velocity
reduced diversity of microhabitat velocities
- fewer protected sites

5. biotic interactions
- competition
- predation
- disease
- parasitism

- increased frequency of diseased **fish**
altered primary & secondary production
- altered trophic structure
- altered decomposition rates and timing
disruption of seasonal rhythms
- shifts in species composition and relative abundances
- shifts in invertebrate functional groups (increased scrapers and decreased shredders)
shifts in trophic guilds (increased omnivores and decreased piscivores)
- increased frequency of hybridization
- increased frequency of non-native species

Figure 5-1. Five major classes of environmental factors that affect aquatic biota. Arrows indicate the kinds of **effects that can be expected from human activities (modified from Karr 1991).**

free of pollutants; high dissolved oxygen concentrations in rearing and incubation habitats; water of low sediment content during the growing season (for visual feeding); clean gravel substrate for reproduction; and unimpeded migratory access to and from spawning and rearing areas.

5.1.1 Food (Energy) Source

As discussed in Section 4.1, salmonids require sufficient energy to meet their basic metabolic needs, to grow, and to reproduce. Maintaining the integrity of aquatic ecosystems depends on maintaining the natural spatial and temporal patterns and amount of primary production. In streams where energy inputs are dominated by allochthonous materials, detrital particles generally are larger than in streams where autochthonous production dominates. In addition, in streams with an intact riparian canopy, the timing and type of material delivered to the channel differs between coniferous and deciduous forests. Together, these factors determine the abundance and species composition of aquatic invertebrates, which are the principal food source for most salmonids. Removal of riparian vegetation in smaller streams changes the dominant energy inputs from allochthonous to autochthonous sources. The conversion of riparian vegetation from conifer-dominated communities to deciduous-dominated communities, or from shrub-dominated to grass-dominated communities, alters the type of food energy available to the system, the temporal patterns of allochthonous inputs, and the invertebrate communities that feed on those resources. Although not all of these changes are necessarily detrimental to salmonids, they represent fundamental changes to ecosystem function. Streams with anadromous fish populations have an additional important source of nutrients in the form of salmon carcasses (see Section 3.8.2), which may contribute substantially to the productivity of the system.

Physical habitat complexity influences the retention and processing of organic materials within streams and rivers. In addition, characteristics of the physical and chemical environment—temperature, streamflow, turbidity, nutrient availability, and physical structure—all influence the composition and abundance of invertebrate communities within streams, lakes, and estuaries, as well as the ability of salmonids to obtain these food resources (see Section 4.1.1). Thus physical and chemical processes must be maintained to ensure that food resources remain within the natural range of abundance for the particular site.

5.1.2 Water Quality

Water temperature, turbidity, dissolved gases (e.g., nitrogen and oxygen), nutrients, heavy metals, inorganic and organic chemicals, and pH all influence water quality and the ability of surface waters to sustain fish populations. With the exception of organic and inorganic chemicals of anthropogenic origin, each of these factors is naturally occurring and exhibits daily or seasonal fluctuations in concentration or magnitude. If the magnitude or concentration of any of these factors exceeds the natural range for a specific location and time of year, biological processes are altered or impaired.

Temperature

Perhaps no other environmental factor has a more pervasive influence on salmonids and other aquatic biota than temperature. The vast majority of aquatic organisms are poikilothermic—their body temperatures and hence metabolic demands are determined by temperature (see Section 4.1). Consequently, virtually all biological and ecological processes are affected by ambient water temperature. Many effects of temperature on these processes have been discussed elsewhere in this document. Below is a brief list of some of the more important physiological and ecological processes affected by temperature, referenced to sections of this document where more detailed discussions may be found

- Decomposition rate of organic materials
- Metabolism of aquatic organisms, including fishes (Sections 4.1.1, 4.1.5)
- Food requirements, appetite, and digestion rates of fishes (Section 4.1.1)
- Growth rates of fish (Section 4.1.1)
- Developmental rates of embryos and alevins (Section 4.1.2)
- Timing of life-history events including adult migrations, fry emergence, and smoltification (Section 4.1.4)
- Competitor and predator-prey interactions (Sections 4.3.2, 4.3.3)
- Disease-host and parasite-host relationships (Section 4.3.4)
- Development rate and life history of aquatic invertebrates

From this list, it is evident that protection and restoration of salmonid habitats requires that temperatures in streams and lakes remain within the natural range for the particular site and season.

Most of the literature on salmonid temperature requirements refers to "preferred", "optimal", or "tolerable" temperatures or temperature ranges (Everest et al. 1985: Bell 1986: Bjornn and Reiser 1991). Preferred or optimal temperatures are generally derived in laboratory studies of behavior (e.g., temperature selection) or performance (e.g., growth, survival, metabolic scope). In general, the term "preferred temperature" is used to describe the temperature to which, given unlimited acclimation time, a fish will ultimately gravitate towards (Fry 1947). The "optimum temperature" means the temperature at which a fish can best perform a specific activity. The "tolerable temperature" range includes temperatures at which fish can survive indefinitely. Although studies of temperature preferences, optima, and tolerances are useful in establishing general physiological requirements, they do not address the ecological requirements of salmonids or local adaptation to specific thermal regimes. For this reason, water-quality criteria that are designed to prevent temperatures from exceeding physiologically stressful levels alone are unlikely to prevent more subtle ecological changes.

Turbidity and Suspended Solids

Turbidity in streams is caused by phytoplankton and by inorganic and organic materials that become suspended during high flow conditions. Inorganic and organic solids enter the aquatic environment in surface runoff, or as particles derived from erosion associated with natural (e.g., slumping of unstable banks, storm runoff, volcanoes) or anthropogenic activities (e.g., forestry, grazing, mining, and agricultural practices) (Leidy 1980; Stumm and Morgan 1981; Dickson et al. 1987; Adriano 1992; Hem 1992).

Turbidity and suspended solids in surface waters can effect periphyton and phytoplankton by reducing light transmission and by causing physical damage through abrasion and scouring (Chandler 1942; Chapman 1963; Bullard 1965; Cairns et al. 1972). A number of studies have indicated that turbidity is a major factor controlling phytoplankton abundance (Buck 1956; Cordone and Pennoyer 1960; Herbert et al. 1961; Benson and Cowell 1967; Sherk et al. 1976). In addition, diminished light penetration and streambed stability can lead to reductions in algal productivity (Samsel 1973) and changes in plant species composition. Samsel (1973) found that a reduction of transparency of about 50% caused a threefold reduction in algal productivity in a Virginia impoundment. Chapman (1963) noted that moving sediment may grind or dislodge algae. Shifting of deposited sand (0.008–0.015 inches) prohibited establishment of periphyton along an English riverbed (Nuttall 1972).

Siltation reduces the diversity of aquatic insects and other aquatic invertebrates by reducing interstices in the substrate. When fine sediment is deposited on gravel, species diversity and densities drop significantly (Cordone and Pennoyer 1960; Herbert et al. 1961; Bullard 1965; Reed and Elliott 1972; Nuttall and Bielby 1973; Bjornn et al. 1974; Cederholm et al. 1978). Deposited sediment may reduce accessibility to microhabitats by embedding the edges of cobbles (Brusven and Prather 1974), and it may also entomb benthic organisms, which then die of oxygen depletion (Ellis 1931). Suspended sediments also limit benthic invertebrates (Tarzwell 1938; Rees 1959; Branson and Batch 1971). In a 10-year stream survey Roback (1962) found numbers of caddis fly larvae genera decreased from 16 to 7 at sediment concentrations in excess of 500 ppm. Addition of more than 80 ppm of inert solids to the normal suspended particle concentration of 40 ppm caused a 60% reduction in population of riffle macroinvertebrates (Gammon 1970). Estuarine copepods ingested fewer food organisms as silt concentration increased (Sherk et al. 1976).

Siltation and turbidity adversely affect fish at every stage of their life cycle (Iwamoto et al. 1978). In general, deposited sediments have a greater impact on fish than do suspended sediments: spawning and incubation habitats are most directly affected (see Section 5.2.2). Particulate materials physically abrade and mechanically disrupt respiratory structures (e.g., fish gills) or surfaces (e.g., respiratory epithelia of benthic macroinvertebrates) in aquatic vertebrates and invertebrates (Rand and Petrocelli 1985). Sediment covers intergravel crevices which fish use for shelter, thereby decreasing the carrying capacity of streams for young salmon and trout (Cordone and Kelley 1961; Bjornn et al. 1974). Fish vacate pools in summer after heavy accumulation of sediments (Gammon 1970). Finally, turbidity affects light penetration, which in turn affects the reactive distance of juvenile and adult salmonids for food capture (see Section 5.2.2).

Although salmonids typically prefer water with low turbidity and suspended sediment content, low levels of turbidity may have beneficial effects. Particulates and dissolved chemical solids, including materials harmful to salmonids, may adsorb to the surfaces of colloidal materials, which in turn can reduce their bioavailability. Thus, adverse effects potentially associated with exposures to inorganic and organic chemicals may be diminished, and biological processes associated with adsorption of dissolved organic solids (e.g., microbial transformation) may enhance the biodegradation and detoxification of organic chemicals in the water (Dickson et al. 1987; Rand and Petrocelli 1985; Adriano 1992; Hem 1992). While adsorption associated with colloids may attenuate adverse biological effects associated with some chemicals, toxicity of other dissolved chemical solids may increase because of interactions with colloidal materials in the water column. The exposure of fish to heavy metals may increase or the solubilization of heavy metals from otherwise insoluble metal compounds may increase in the presence of suspended solids having a high colloidal content (Leidy 1980; Rand and Petrocelli 1985; Brown and Sadler 1989).

Dissolved Oxygen and Nitrogen Gases

All salmonids require high levels of dissolved oxygen (DO). Reduced levels of oxygen can affect the growth and development of embryos and alevins, the growth of fry, and the swimming ability of adult and juvenile migrants. In most natural situations, DO levels are sufficient to allow normal function, but concentrations may be reduced by large amounts of organic debris, nutrient enrichment from sewage treatment plants and agricultural runoff, and excessively high temperatures. Bjornn and Reiser (1991) reviewed a number of papers and concluded that while thresholds for survival are generally low (3.3 mg/L), growth and food conversion efficiency are affected at DO levels of 5 mg/L, and that DO levels of 8–9 mg/L or more are needed to ensure that normal physiological functions of salmonids are not impaired. EPA's water-quality criteria for dissolved oxygen are 9.5 mg/L for a 7-day mean and 8.0 mg/L for a l-day minimum (EPA 1986). Supersaturation of oxygen gas may occur associated with spills from dams or highly turbulent waters. The EPA standard for maximum levels of oxygen is 110% of normal saturation. **A** more detailed discussion of specific oxygen requirements at each life stage is presented in Section 5.2.

Numerous studies of nitrogen supersaturation indicate that dissolved nitrogen generally affects fish when saturation exceeds 110%–130%, with the threshold level depending on water depth (Rucker and Tuttle 1948; Harvey and Cooper 1962; Fickeison et al. 1973; Blahm 1974; Meekin and Allen 1974; Meekin and Turner 1974; Rucker and Kangas 1974; Blahm et al. 1975; Dawley et al. 1975; Weitkamp 1975; Bentley and Dawley 1976; Bouck et al. 1976; Nebeker and Brett 1976). Gas bubble disease (GBD) and mortality are the primary detrimental effects associated with dissolved nitrogen concentrations above threshold levels (Parametrix 1975).The detrimental effects of nitrogen supersaturation vary according to the length of exposure (Blahm et al. 1975). Chinook salmon, coho salmon, steelhead trout, rainbow trout, whitefish, and largemouth bass were exposed to nitrogen levels of 130% for 8 of every 24 hours. Mortality did not exceed 50% if fish were placed in nitrogen-saturated water (i.e., 100%) of the remaining 16 hours. However, when fish were continuously exposed to supersaturated levels of nitrogen (130%), mortality rates exceeded 50% during the first day. Various species of juvenile salmonids may compensate for total nitrogen saturation levels up to 125% by remaining in deeper water (Parametrix 1975). Hydrostatic pressure increases with depth, so in deeper water nitrogen remains in solution in the blood of fish, inhibiting GBD.

Nutrients

Nutrient levels should remain within the natural range for the area and season as well as sustain the normal level of primary production. Various inorganic constituents of surface water are nutrients required for biological processes. Nitrogen and phosphorus are clearly the most important nutrients affecting productivity of aquatic systems. Natural sources of nitrogen and phosphorous in natural ecosystems are discussed at length in Section 3.8.1. Inputs to surface- and groundwaters can be affected by vegetation changes associated with land-use activities as well as through direct enrichment from sewage effluents, run-off from agricultural lands, and industrial water.

Nitrogen generally occurs in natural waters as nitrite or nitrate anions, as cationic forms like ammonium, and as intermediate oxidation states like those that occur in biological materials (e.g., decomposing organic solutes). In surface waters or in groundwaters that are impacted through human use, cyanide from industrial sources, mines, and various other complex effluents (including agricultural runoff) may also be sources of nitrogen. Nitrite-nitrogen is short-lived in natural environments and, consequently, rarely exists in concentrations toxic to salmonids. Acute toxicity values for salmonids range from 100 to 900 ppb as NO_2-N (48-hour or 96-hour LC_{50}); chronic effects are poorly understood, in part, because nitrite does not persist in surface waters under natural conditions.

Nitrate is formed by the complete oxidation of ammonia through the nitrification process and can be found in relatively high concentrations in surface waters. Unlike ammonia and nitrite, nitrate does not form un-ionized species in aqueous solutions and is considered essentially nontoxic for aquatic vertebrates and invertebrates (e.g., acute LC„ greater than 1300 ppm for salmonids). However, much lower concentrations of nitrate may lead to adverse effects associated with eutrophication and the development of oxygen-depleted waters (Leidy 1980; Rand and Petrocelli 1985).

Ammonia frequently acts as a toxicant in surface waters subject to high inputs of nitrogen, especially through anthropogenic activities (e.g., agricultural runoff, sewage effluents). For salmonids, ammonia is acutely toxic at concentrations as low as 80 ppb, but the initiation of ammonia toxicosis is highly variable, primarily as a function of pH. Physiological responses to ammonia exposure are frequently exacerbated by low dissolved oxygen concentrations; for salmonids, acute toxicity is increased two-fold when dissolved oxygen is decreased from 80% to

30% saturation. In the laboratory, chronic effects of ammonia have been documented as low as 2 ppb, but little work has been completed to identify the effects of long-term exposures under field settings (Rand and Petrocelli 1985; Reader and Dempsey 1989).

In contrast to nitrogen, phosphorus does not leach as readily from soil. In natural waters, phosphorus occurs in very low concentrations, most often in tenths of a milligram per liter (or less). Orthophosphate and its intermediates most frequently occur in surface waters and are routinely measured as "total phosphorus" in water-quality monitoring activities. Phosphorus most frequently occurs in surface waters as phosphates, which are generally considered nontoxic to aquatic vertebrates and invertebrates (Stumm and Morgan 1981; EPA 1986).

Nitrogen and phosphorus are the principal causes of nutrient enrichment of surface waters. Aquatic macrophytes (rooted-submerged and floating vascular plants) and algae are dependent to varying degrees on dissolved nitrogen and phosphorus for their nutrient supply. Growth of benthic algae and phytoplankton is particularly sensitive to the ratio of nitrogen to phosphorus. Enrichment leads to high production rates of biomass (e.g., algal blooms) that are undesirable for other aquatic biota, especially when respiration and decomposition create high biochemical oxygen demand and oxygen depletion. While the enhanced growth rates of aquatic vegetation can reach maximal conditions under nitrogen and phosphorus enrichment, phosphorus frequently acts as the limiting factor in aquatic habitats and will tend to control production rates (Leidy 1980; Stumm and Morgan 1981; Hem 1992).

Biocides

Agricultural chemicals are potentially widespread in the environment, and surface waters and groundwaters may be affected by chemical use that accompanies changes in land-use practices. Various classes of chemicals are currently used in the agricultural industry, including herbicides, insecticides, fungicides, nematicides, defoliants, rodenticides, and growth regulators. These are primarily organic chemicals, but inorganic chemicals, such as mineral salts and nutrients, may also be used as fertilizers and may directly effect receiving waters. Similarly, complex chemical mixtures from industries, municipalities, and landfills may impact water resources through runoff or infiltration to groundwater (Leidy 1980; Rand and Petrocelli 1985).

Agricultural chemicals are regulated to decrease the likelihood of their release into surface waters and groundwaters, and water-quality criteria have been established for many of these chemicals (Table 5-1). There are several properties of organic chemicals that influence their fate and effects in the environment. For surface waters and groundwaters, a chemical's adsorptivity, stability, solubility, and toxicity will determine the extent to which that chemical will migrate and adversely effect a water resource. Among the thousands of agricultural chemicals available for users (industries, small businesses, farmers, orchardists, and home gardeners), these properties will vary significantly. Depending upon the chemical's physicochemical properties, the potential contamination of water resources may be complex. For example, a chemical's water solubility will influence whether it occurs in solution or adsorbed to sediments or colloids held in suspension (Dickson et al. 1987; Rand and Petrocelli 1985). Synthetic organics, even at subacute levels, may alter neurological, endocrine, and behavioral functions in fish (e.g., Folmar 1993; Choudhary et al. 1993; Singh et al. 1994). In addition to being toxic to fish and invertebrates that fish eat, organic chemicals may indirectly affect nontarget species through habitat alteration (e.g., changes in plant community structure as a result of targeting weedy species), and such changes may occur even under the best management practices (Leidy 1980).

Heavy Metals

Metal concentrations in surface water vary regionally, reflecting the geochemical composition of the underlying parent material and the soils characteristic of the watershed. Most frequently, metals occur in trace quantities as a result of soil leaching and geochemical processes that occur in the underlying bedrock. The concentration of metals in surface waters may be increased by anthropogenic activities such as mining and related industrial practices, such as electroplating and metals refining (Leidy 1980; Stumm and Morgan 1981; Rand and Petrocelli 1985).

Although some metals are necessary trace nutrients, many metals are toxic to fish at very low concentrations. Other water-quality conditions influence the bioavailability of the metals. For example, metals that are nutritional requirements must be absorbed by the organism. Metals may occur in solution and may be available for uptake directly from the water, or they may be adsorbed to colloidal particles in the water column. The extent to which metals are adsorbed and then intentionally or coincidentally ingested may influence the onset of metal toxicosis in aquatic biota, especially when the interaction between the metals in solution and metals adsorbed to colloids of various forms (e.g., relatively simple organic ligands versus complex organic structures like the humic acids) is influenced by other water-quality conditions such as pH. Table 5-2 lists water-quality criteria for selected metals and metalloids that are frequently considered toxicants of

Table 5-1. Water-quality criteria for selected herbicides, pesticides, and fungicides in freshwaters. From EPA (1986).

Chemical	Maximum acceptable levels ($\mu g/L$)	
	Acute (instantaneous)	Chronic (24-hour average)
Aldrin	3.0	
Chlordane	2.4	0.0043
Chlorophenoxy herbicides		
2,4-D	100.0	
2,4,5-TP	10.0	
Chloropyrifos	0.083*	
DDT and metabolites		
DDT	1.10†	0.001†
TDE	0.60†	
DDE	1050.0†	
Dieldrin	2.5	0.0019
Endosullfan	0.22	0.056
Endrin	0.18	0.023
Guthion	0.01	
Heptachlor	0.52	0.0038
Lindane	2.0	0.080
Malathion	0.10	
Methoxychlor	0.012	0.030
Mirex	0.001	
Parathion	0.04	
Pentachlorophenol	‡	
Toxaphene	1.60	0.013

* l-hour average, not more than 1 time per 3 years.
† Human-health based criteria.
‡ Criteria based on pH: see current EPA criteria.

Table 5-2. Water-quality criteria for metals and metalloids found in surface waters. Criteria for most metals are a function of water hardness. Also shown are other factors affecting toxicity and anthropogenic sources. From EPA (1986).

Metal	Anthropogenic source	Maximum acceptable levels (μg/L)		Other factors influencing toxicity
		4-day average*	l-hour average*	
Arsenic	Agrichemicals	190	360	
Boron	Agrichemicals	750[†]		
Cadmium	Mininglindustrial	$e^{(1\ 128[\ln(\text{hardness})]-3.49)}$	$e^{(1.128[\ln(\text{hardness})]-3.828)}$	pH
Chromium VI	Industrial	11	16	pH
Chromium III	Industrial	$e^{(0.819[\ln(\text{hardness})]+1.561)}$	$e^{(0.819[\ln(\text{hardness})]+3.688)}$	pH
Copper	Mininglindustrial	$e^{(0.8545[\ln(\text{hardness})]-1.465)}$	$e^{(0.9422[\ln(\text{hardness})]-1.464)}$	pH, valence, temperature, other metals
Iron	Mininglindustrial		1,000[#]	
Lead	Mininglindustrial	$e^{(1.273[\ln(\text{hardness})]-4.705)}$	$e^{(1.273[\ln(\text{hardness})]-1.460)}$	Turbidity, pH
Manganese	Industrial	50[‡]		
Mercury	Industrial	0.144[§]		DOC, microbial activity
Nickel	Mining/industrial	$e^{(0.76[\ln(\text{hardness})]+1.06)\P}$	$e^{(0.76[\ln(\text{hardness})]+4.02)\#}$	
Selenium	Irrigated agriculture	35[¶]	260[#]	
Silver	Mining		$e^{(1.72[\ln(\text{hardness})]-6.52)\#}$	
Zinc	Mininglindustrial		$e^{(0.83[\ln(\text{hardness})]+1.95)\#}$	pH, temperature, valence

* Values not to be exceeded more than 1 time per 3 years.

[†] Criteria is for long-term irrigation of crops. No freshwater standards given

[‡] Domestic water supplies.

[§] Human health criteria.

[¶] 24 hour average.

[#] Instantaneous.

concern, especially in surface waters impacted by human use. The physical features of the surrounding habitat (e.g., land-use in riparian areas) may influence a chemical's toxicity, and seasonal variations in bioavailability of contaminants (e.g., changing redox potentials of sediments and availability of metals) must also be considered (Stumm and Morgan 1981; Dickson et al. 1987; Adriano 1992).

pH

Acidic surface waters may occur naturally as a result of dissolution of parent materials in bedrock and overlying soils, biological decomposition (especially processes yielding organic acids such as fulvic and humic acids), or through geothermal activity or catastrophic events related to volcanic activity. More frequently however, surface-water acidity results from anthropogenic activities related to land use (e.g., mining) or resource use (e.g., combustion of fossil fuels) with the subsequent deposition of materials capable of generating—directly or indirectly—and releasing hydrogen ions to the environment (Leidy 1980; Rand and Petrocelli 1985). The influence of hydrogen ions on aquatic organisms is influenced by watershed characteristics, including the buffering capacity of soils as well as by concentrations of dissolved materials in surface waters (Rand and Petrocelli 1985; Brown and Sadler 1989).

In general, fish may be adversely affected by surface water with pH 5.6 or less; however, the threshold for adverse effects is species-specific and water-quality dependent (e.g., buffering capacity). Hence, no single pH value can be regarded as a threshold for anticipating population-level responses to acidic surface waters. Respiratory problems are frequently observed in experimental fish exposed to low pH. Mucous clogging, increased ventilation, coughing and hypoxia are commonly recorded in acid-exposed fish. Aluminum and other metals exacerbate the physiological response to increased hydrogen ion. Low pH alters the specific form of metals in soils, increasing both their mobility and their bioavailability to aquatic organism. In addition, low pH acts synergistically with heavy metals in surface waters to yield adverse biological effects (Stumm and Morgan 1981; Rand and Petrocelli 1985; Brown and Sadler 1989). High pH values may also adversely affect salmonids. Elevated pH can arise when reductions in canopy cover in riparian zones stimulates production of algae. As algae photosynthesize during the day, they take up carbon dioxide, which results in a reduction in free hydrogen ions (i.e., increasing pH).

5.1.3 Habitat Structure

The physical structure of streams, rivers, and estuaries plays a significant role in determining the suitability of aquatic habitats to salmonids as well as other organisms upon which salmonids depend for food. These structural elements are created through interactions between natural geomorphic features, the power of flowing water, sediments that are delivered to the stream channel, and riparian vegetation, which provides bank stability and inputs of large woody debris. Structural attributes of streams vary naturally among regions and along the longitudinal dimension of streams in response to differences in topography, geology, geomorphic features, hydrologic regime, sediment load, and riparian vegetation (see Sections 3.5 and 3.9.5). These spatial differences and gradients give rise to the variety of macro- and microhabitat attributes that are used by salmonids at various stages of their life histories. Macrohabitat features include pools, glides, and riffles. The relative frequency of these habitat types changes with size of the stream, the degree of channel constriction, and the presence of large woody debris. Microhabitat attributes include characteristics such as substrate type, cover, depth, hydraulic complexity, and current velocity.

Because of the great diversity in the physical attributes of western streams and in the requirements of various salmonids, and because few undisturbed watersheds remain to serve as reference points, it is difficult to quantify natural ranges of physical habitat features in streams, rivers, and estuaries. For example, historically, mid-order streams west of the Cascade crest had 16–38 pools per km (25–60 per mile) (FEMAT 1993). Pool frequencies in 10 human-influenced tributaries of the upper Grande Ronde River ranged from 3.8–26.2 per km (6–42 per mile) in 1941 and 1.4–7.4 per km (2–12 per mile) in 1990 (McIntosh et al. 1994b). In the Yakima Basin, an unmanaged watershed (Rattlesnake Creek) averaged 1.6 pools per km (2.5 per mile) in 1935–1936 and 3.9 pools per km (6 per mile) in 1987–92 (McIntosh et al. 1994b); similar pool frequencies were also reported for the Chewack River in the Methow River Basin. In low gradient streams on the Olympic Peninsula, Washington, pools constituted 81.1% of the stream surface area (160 pools $> 10\ m^2$ per mile) (Grette 1985). In low-gradient stream reaches in southeast Alaska, pools accounted for 39%–67% of the surface area depending upon bank full width (Murphy et al. 1984 discussed in Peterson et al. 1992). This high degree of variation illustrates the importance of local geomorphic features, stream size, and riparian influence on stream habitat characteristics.

Despite inherent differences in streams, it is clear that habitat complexity is an important feature of aquatic systems. In streams of the Pacific Northwest, large woody debris creates both macro- and microhabitat complexity that is essential to salmonids and other aquatic organisms. Large wood creates habitat heterogeneity by forming pools, back eddies, and side channels and by creating channel sinuosity and hydraulic complexity. Large wood also functions to retain coarse sediments (e.g., spawning gravels) and organic matter in addition to providing substrate for numerous aquatic invertebrates. McIntosh et al. (1994b) reported that changes in substrate composition towards smaller fractions coincided with reduced frequency of large woody debris in streams of the upper Grande Ronde River. Consequently, large woody debris plays a significant part in controlling other structural elements of streams.

Large woody debris provides an important component to estuarine habitats of coastal rivers (Maser et al. 1988), which are important rearing areas for juvenile anadromous salmonids (Table 4-2). Woody debris increases habitat complexity in areas where the bottom consists mainly of fine sediments. Numerous invertebrates rapidly process the wood, liberating nutrients for some organisms, while others use the wood as refugia. In salt marshes, large woody debris traps sediments to increase the extent of the marsh. As exceptionally high tides displace the logs, depressions left in the sediments increase habitat diversity important to juvenile fishes. In areas that are predominantly mud bottomed, large wood further serves as a repository for herring spawn. The functional roles of large woody debris in streams, and how these change from headwater reaches to estuaries, are reviewed in greater detail in Section 3.9.5. The functions of large woody debris relative to specific life stages of salmonids are discussed in Section 5.2.

Other important components of habitat structure at the microscale include large boulders, coarse substrate, undercut banks, and overhanging vegetation. These habitat elements offer salmonids concealment from predators and shelter from fast currents. At the macrolevel, streams and rivers with high channel sinuosity, multiple channels and sloughs, beaver impoundments, or backwaters typically provide high-quality habitat for salmonids. Such areas serve as refugia during high flows. Salmonids in estuaries benefit from similar structural features (substrate complexity, overhanging vegetation, depth heterogeneity) as well as abundant macrophytes.

5.1.4 Flow Regime

Flow regimes in streams and rivers determine the amount of water available to salmonids and other aquatic organisms, the types of micro- and macrohabitats that are available to salmonids (see Section 5.2), and the seasonal patterns of disturbance to aquatic communities. High-flow events redistribute sediments in streams, flushing fine sediments from spawning gravels and allowing recruitment of gravels to downstream reaches. In addition, extreme flow events are essential in the development and maintenance of healthy floodplain systems through deposition of sediments, recharge of groundwater aquifers, dispersal of vegetation propagules, recruiting large woody debris into streams, and transporting wood downstream. In alluviated reaches, high flows may create new side channels and flood off-channel areas that are important rearing habitats for salmonids. Low flow may also be important for the establishment of riparian vegetation on gravel bars and along stream banks (Section 3.6). Thus, although over shorter time scales high- or low-flow events may temporarily reduce salmonid numbers, dynamic flows are needed to perform essential functions important in the long-term persistence of salmonid populations.

The specific flow requirements of salmonids vary with species, life history stage, and time of year (see Section 5.2). Local salmonid populations have evolved behavioral and physical characteristics that allow them to survive the flow regimes encountered during each phase of their development. Protection of salmonid habitats requires streamflows to fluctuate within the natural range of flows for the given location and season.

5.1.5 Biotic Interactions

Protecting and restoring biological integrity in surface waters also depends on maintaining natural biological interactions among species. These interactions may be affected directly by the introduction of non-native species and stocks (see Sections 4.3 and 6.12) and overexploitation (Section 6.11) or indirectly through modification of physical and chemical characteristics of streams, lakes, and estuaries (reviewed in Chapter 4). Human-induced impacts on biological interactions include changes in primary and secondary production, disruption in timing of life history events or seasonal rhythms, increased frequency of disease or parasitism, and changes in the outcome of predator-prey and competitive interactions. Together these perturbations lead to changes in food webs and trophic structure of aquatic systems.

5.2 Habitat Requirements by Life Stage

Salmonids use a variety of habitats during their life histories. Anadromous species in particular have complex life histories that involve periodic shifts in habitat. Depending on the species or stock, freshwater streams, lakes, or intertidal sloughs may be used for reproduction; streams, lakes, estuaries, or oceans may be used for juvenile rearing (Table 4-2). For all anadromous species, habitats between spawning streams and the ocean are required for upstream and downstream migrations. Differences in spatial and temporal use of specific habitats exist for each species, yet the diversity among species and by life stage indicates that most freshwater habitats are utilized year round (Table 4-3). Juvenile-to-adulthood rearing generally occurs in the ocean, but there is considerable variation (Table 4-2), even within each species. To persist, each species or stock must be able to survive within the entire range of habitats encountered during its life; degradation or alteration of habitat required at any life stage can limit production. Much of the available information on salmonid habitat requirements has been summarized in reviews by Bell (1986), Everest et al. (1985), and Bjornn and Reiser (1991), which are the primary sources of information for this section unless otherwise noted.

Most of the quantitative descriptions of requirements for salmonid habitats presented in this section consist of either microhabitat observations of salmonids in nature or results from laboratory studies that measure the performance of salmonids (often hatchery fish) under controlled conditions. Microhabitat measurements are frequently made during a single season (usually summer, when sampling is easiest), and the resulting data are often reported in the literature without accompanying data on habitat availability. Habitat utilization constitutes a "preference" only when the particular range of depths, velocities, substrates, or cover types is used at a frequency greater than its general availability in the environment. The range of microhabitats used in a particular stream depends on availability. Consequently, variation in microhabitat utilization among streams (or segments of the same stream) can be substantial, and habitat preference information derived from one location should not be applied to other areas without careful consideration of similarities and differences between sites. In addition, microhabitat measurements at holding positions of salmon and trout do not always encompass the range of velocities needed for feeding, which are commonly higher. For all of these reasons, care must be taken when interpreting microhabitat data published in the literature. Similarly, optimal conditions for development, growth, and survival as determined in the laboratory do not always correspond to the most favorable conditions in natural environments (see Section 5.1).

5.2.1 Adult Migrations

The migrations of anadromous salmonids from river mouths to their natal streams vary in length from a few hundred meters (e.g., chum salmon spawning in the intertidal zone) to well over a thousand kilometers. Even resident fish may make substantial migrations between lakes and streams or between sections of a river network (Everest et al. 1985). During upstream migrations, anadromous salmonids need holding or resting sites and suitable flow and water quality. Resident species may feed during their migrations and thus may have more diverse habitat needs.

Physical Structure

Upstream migration of many salmonid species typically involves rapid movements through shallow areas, followed by periods of rest in deeper pools. Some races, such as spring chinook and summer steelhead, may arrive at spawning sites several months before spawning or hold in mainstem rivers for several weeks or months prior to moving into their natal streams to spawn (Bjornn and Reiser 1991). Large woody debris, boulders, and other structures provide hydraulic complexity (e.g., eddies or localized areas of slow water) and pool habitats that serve as resting stations for fish as they migrate upstream to spawn. Resident species use structure to pause out of the main current while waiting for prey to drift by in adjacent, faster waters. Large woody debris and other structures may also facilitate temperature stratification and the development of thermal refugia by isolating pockets of cold water and preventing mixing (see discussion of temperature below). In shallower reaches, riparian vegetation and large wood provide cover from terrestrial predators. At redd sites, adequate areas of stable, appropriately sized gravel containing minimal fine sediments are required for successful spawning (see Section 5.2.2).

Flows and Depth

Streamflow during the spawning migration must be sufficient to allow passage over physical barriers including falls, cascades, and debris jams; as a result, the migrations of many stocks occur coincident with high flows. Coho salmon frequently wait near stream mouths until a freshet occurs before moving upstream (Sandercock 1991), as may pink salmon (Heard 1991). Holtby et al. (1984) observed continuous entry of coho salmon into Carnation Creek during years of high flow but pulsed entry when freshets were infrequent. Spring and summer chinook stocks migrate during periods of high flows that allow them to reach spawning tributaries in headwater reaches, while fall-run stocks, which typically spawn in lower reaches, may enter streams during periods of relatively low flow (Healey 1991).

Minimum depths that will allow passage of salmonids are approximately 12 cm for trout, 18 cm for the smaller anadromous species (i.e. pink, chum, steelhead, sockeye, and coho salmon), and 24 cm for large chinook salmon (Bjornn and Reiser 1991); however, substantially greater depths may be needed to negotiate larger barriers. Reiser and Peacock (1985) report that maximum leaping ability varies from 0.8 m for brown trout to 3.4 m for steelhead. Pool depths must exceed barrier height by approximately 25% to allow fish to reach the swimming velocities necessary to leap to these heights (Stuart 1962). The ability to pass a barrier is also influenced by pool configuration. Water plunging over a steep fall forms a standing wave that may allow salmonids to attain maximum heights (Bjornn and Reiser 1991). Less severe inclines (e.g., cascades) may be more difficult to pass if pool depths are inadequate and velocities are high.

Water Quality

Temperature. Most adult salmonids typically migrate at temperatures less than 14"C; however, spring and summer chinook salmon migrate during periods when temperatures are substantially warmer (Table 5-3). Excessively high or low temperatures may result in delays in migration (Major and Mighell 1966; Hallock et al. 1970; Monan et al. 1975). Adult steelhead that move from the ocean into river systems in the summer and fall may overwinter in larger rivers, delaying entry into smaller spawning tributaries until they are free of ice in the spring. Similarly, spring-spawning resident salmonids, including cutthroat and rainbow trout, may hold at the mouths of spawning streams until temperatures warm up to the preferred temperature range (Bjornn and Reiser 1991). In addition to delaying migration, excessively high temperatures during migration may cause outbreaks of disease (see Section 4.3.4).

Coldwater refugia may also be important to adult salmon as they migrate upstream. Adult summer-run steelhead in the Middle Fork Eel River of California were observed in thermally stratified pools, but were absent or infrequent in non-stratified pools of similar depth (Nielsen et al. 1994). Coldwater pockets in stratified pools ranged from 4.1 to 8.2°C cooler than ambient stream temperatures. Spring chinook salmon have also been observed to hold in coldwater pools for several months prior to spawning in the Yakima River of eastern Washington, moving as much as 60 km from holding pools to spawning sites (NRC 1992). The authors suggest that this behavioral thermoregulation lowers metabolic rates and thereby conserves energy for gamete production, mate selection, redd construction, spawning, and redd guarding.

Streamflow, channel morphology, and the presence of large woody debris may play significant roles in mediating the formation and persistence of coldwater refugia (Bilby 1984; Nielsen et al. 1994). In some streams and rivers, gravel bars or other structures isolate incoming tributaries or seep areas from mainstem waters, thereby inhibiting the mixing of waters and helping to maintain thermal gradients (Nielsen et al. 1994). In larger systems, thermally stratified pools need not be associated with coldwater inputs provided that deep scour pools exist and flows are sufficiently low to prevent turbulent mixing. Consequently, in larger systems management practices that reduce large woody debris, destabilize stream channels, increase turbulence or modify stream flows may eliminate coldwater refugia.

Dissolved Oxygen. The high energy expenditures of sustained upstream swimming by salmonids requires adequate concentrations of dissolved oxygen (DO). Davis et al. (1963) found adult and juvenile swimming performance impaired when DO dropped below 100% saturation levels for water temperatures between 10–20°C. DO concentrations below 6.5–7.0 mg/L greatly impaired performance at all temperatures studied. Migrating adults exhibited an avoidance response to DO levels below 4.5 mg/L (Hallock et al. 1970). Migration resumed when DO levels increased to 5 mg/L.

Turbidity. High concentrations of suspended sediment may delay or divert spawning runs and in some instances can cause avoidance by spawning salmon (Smith 1939; Servizi et al. 1969; Mortensen et al. 1976). Salmonids were found to hold in a stream where the suspended sediment load reached 4,000 mg/L (Bell 1986). Though high sediment loads may delay migration, homing ability does not seem to be adversely affected (Murphy 1995). Cowlitz River chinook salmon returned to the hatchery seemingly unaffected by the sediments derived from the eruption of Mount St. Helens, Washington, although in the highly impacted Toutle River tributary of the Cowlitz, coho salmon did stray to nearby streams for the first two years following the eruption (Quinn and Fresh 1984).

Table 5-3. Tolerable and preferred temperature ranges (°C) for adult migration, spawning, and incubation of embryos for native salmonids in the Pacific Northwest. Modified after Bjornn and Reiser (1991).

	Life Stage		
Species	Spawning Migration (min – max)	Spawning (preferred range)	Incubation (preferred range)
ANADROMOUS			
Pink salmon	7.2 – 15.6*	7.2 – 12.8*	4.4 – 13.3*
Chum salmon	8.3 – 15.6*	7.2 – 12.8*	4.4 – 13.3*
Coho salmon	7.2 – 15.6*	4.4 – 9.4*	4.4 – 13.3*
Sockeye salmon	7.2 – 15.6*	10.6 – 12.2*	4.4 – 13.3*
Spring chinook	3.3 – 13.3''	5.6 – 13.9*	5.0 – 14.4*
Summer chinook	13.9 – 20.0*	5.6 – 13.9*	5.0 – 14.4*
Fall chinook	10.6 – 19.4*	5.6 – 13.9*	5.0 – 14.4*
Steelhead trout		3.9 – 9.4*	
Cutthroat trout		6.1 – 17.2*	
RESIDENT			
Kokanee		5.0 – 12.8*	
Mountain whitefish		0.0 – 5.6†	
Cutthroat trout	5.0 – 10.0	4.4 – 12.8† 5.5 – 15.5‡	
Rainbow trout		2.2 – 20.0* 4.4 – 12.8†	
Dolly Varden		7.8†	
Bull trout		< 9.0§ 4.5¶	2.0 – 6.0§

* Bell 1986.
† Everest et al. 1985.
‡ Varley & Gresswell 1988.
§ Pratt 1992.
¶ Ratliff 1992.

5.2.2 Spawning and Incubation

Although spawning and incubation occur in the same habitat, adults and embryos have slightly different habitat needs. Adults select sites based on substrate composition, cover, and water quality and quantity. Embryo survival in and fry emergence from an intragravel environment depends upon physical, hydraulic, and chemical variables including substrate size, channel gradient and configuration, water depth and velocity, DO, water temperature, biochemical oxygen demand in the gravel, and permeability and porosity of the gravel in the redd (Bjornn and Reiser 1991).

Physical Structure

All salmonids require sufficient gravels within a specific size range and a minimum of fine sediments for successful spawning. Usable gravel size generally is proportional to adult size—larger individuals spawn in larger substrate (Marcus et al. 1990). Bjornn and Reiser (1991) reviewed the available literature and found that anadromous salmon typically use gravels in the 1.3–10.2-cm size range, whereas steelhead and resident trouts may use smaller substrates (0.6–10.2 cm). The depth that salmonids deposit eggs within the substrate is also a function of size (Everest et al. 1985) and may be critical to incubation success. Nawa and Frissell (1993) found that gravel beds can be both scoured and filled during the same flood event potentially leaving little net change in bed surface elevation. Eggs deposited within the zone of scour and fill are likely to wash downstream. Bedload and bank stability arising from LWD and intact upslope, floodplain, or riparian zones minimize this risk. Large woody debris diversifies flows, reducing stream energy directed towards some portions of the stream (Naiman et al. 1992). This creates pockets of relatively stable gravels better protected from the scouring effects of high flows.

Flow and Depth

The number of spawning salmon and trout that can be accommodated in a given stream depends on the availability of suitable habitats for redd construction, egg deposition, and incubation (Bjornn and Reiser 1991). Two characteristics of spawning habitats directly tied to streamflow are water depth and current velocity. Salmonids typically deposit eggs within a range of depths and velocities that minimize the risk of desiccation as water level recedes and that ensure the exchange of water between surface and substrate interstices is adequate to maintain high oxygen levels and remove metabolic wastes from the redd. In general, the amount of habitat suitable for spawning increases with increasing streamflow; however, excessively high flows can cause scouring of the substrate, resulting in mortality to developing embryos and alevins (Hooper 1973).

Bjornn and Reiser (1991) recently reviewed studies quantifying specific water depths and velocities at sites used by salmonids for spawning in rivers and streams. In Table **5-4,** results from their review have been supplemented with data from four other reviews (Healey 1991; Heard 1991; Salo 1991; and Sandercock 1991) on spawning sites for anadromous salmonids. Usually, depth and velocity of water at spawning sites is related to the size of spawners: larger species spawn at greater depths and faster water velocities than smaller species. There is also substantial variation among rivers, probably reflecting differences in habitat availability. Most species typically spawn at depths greater than 15 cm, with the exception of kokanee salmon and smaller trout (Table 5-4), which spawn in shallower waters. Location of redd sites based on water depths and velocities may also vary depending on spawner density. For example, pink salmon tend to spawn in shallower waters when conditions are crowded or streamflow is low (Heard 1991). Several species of salmonids may seek out areas of upwelling for spawning; these include sockeye salmon, chum salmon, coho salmon, and bull trout (Burgner 1991; Salo 1991; Sandercock 1991; Pratt 1992). Upwelling increases circulation of water through redds, which helps to eliminate wastes and prevents sediments from filling in spawning gravel interstices. Thus infiltration that recharges groundwater, which eventually discharges in subsurface springs and seeps, must be maintained.

Water Quality

Temperature. Salmonids have been observed to spawn at temperatures ranging from 1–20°C (Bjornn and Reiser 1991), but most spawning occurs at temperatures between 4 and 14°C (Table 5-3). Resident trouts, including rainbow and cutthroat trout, may spawn at temperatures up to 20.0°C and 17.2°C, respectively, while coho salmon, steelhead trout, Dolly Varden, bull trout, and mountain whitefish tend to prefer lower temperatures. The wide range of spawning temperatures utilized by most salmonid species strongly suggests that adaptation has allowed salmonids to persist in a variety of thermal environments and that attempting to identify species-specific preferenda may fail to account for ecological requirements of individual stocks.

Among the salmonids, the preferred incubation temperatures have been best documented for the anadromous species. Bell (1986) suggested preferred temperature ranges of 4.4–13.3°C for pink salmon, chum salmon, coho salmon, and sockeye salmon, and 5.0–14.4°C for chinook salmon (Table 5-3). More

Table 5-4. Water depths and velocities used by anadromous and resident salmonids for spawning.

Species	Depth (cm)	Velocity (cm/s)	Source
Chinook salmon (race not specified)	15–43*	37–69*	Bovee (1978)
	52–128†	55–1137	Graybill et al. (1979)
	30–460		Chapman (1943)
Spring chinook salmon	≥ 24	30–91	Thompson (1972)
	18–38*	24–61*	Bovee (1978)
	5–122		Burner (1951)
	13–720	30–150	Vronskiy (1972)
	45–52	52–68	Collings et al. (1972)
		22–64	Smith (1973)
	30–107	30–53	Chambers et al. (1955)
		15–100	Neilson and Banford (1983)
Summer chinook salmon	≥ 30	32–109	Reiser and White (1981)
	5–700	10–189	Healey (1991)
Fall chinook salmon	10–120	25–115	Bovee (1978)
	≥ 24	30–91	Thompson (1972)
	122–198	84–114	Chambers et al. (1955)
	28–41	30–76	Briggs (1953)
	30–45	30–68	Collings et al. (1972)
		19–81	Smith (1973)
	to 700	37–189	Chapman et al. (1986)
Chum salmon	≥ 18	46–101	Smith (1973)
	13–50†	21–84†	Johnson et al. (1971)
	20–110	10–20	Sano and Nagasawa (1958)
	30–100	10–100	Soin (1954)
Coho salmon	≥ 18	30–91	Thompson (1972)
	4–33	30–55	Gribanov (1948)
	12–35*	25–61*	Bovee (1978)
	20–25	25–70	Li et al. (1979)
	10–20	30–75	Briggs (1953)
Pink salmon	≥ 15	21–101	Collings (1974)
	10–150	30–140	Heard (1991)
Sockeye salmon	≥ 15	21–101‡	Bjornn and Reiser (1991)
	15–300		Burgner (1991)
	17–49††	34–58*	Bovee (1978)
	15–55‡‡	28–79‡‡	Stober and Graybill (1974)
	30–46	53–55	Clay (1961)
Kokanee salmon	≥ 6	15–73	Smith (1973)
	6–23*	11–41*	Bovee (1978)
Steelhead trout (race not specified)	≥ 24	40–91	Smith (1973)
	18†	30–91†	Stober and Graybill (1974)
	12–70	37–109	Hunter (1973)
	27–88t	46–91†	Graybill et al. (1979)
Winter Steelhead trout	24–55*	43–87*	Bovee (1978)
Rainbow trout	≥ 18	48–91	Smith (1973)
	15–43	27–79	Chambers et al. (1955)
	21–30	30	Li et al. (1979)
Cutthroat trout	≥ 6	11–72	Hunter (1973)
	17–30	15–46	Chambers et al. (1955)
Mountain whitefish	≥ 23	30–66*	Bovee (1978)
	610–1220	≥ 15	Li et al. (1979)

* Values indicate 50% probability range
† Values indicate 80% probability range.
‡ Estimated by Bjornn and Reiser (1991) based on criteria for other species.

recent laboratory studies have demonstrated that coho and sockeye salmon embryos tend to be less sensitive to cold temperatures and more sensitive to warm temperatures than pink, chum, or chinook salmon (Murray and McPhail 1988; Beacham and Murray 1990). Coho and sockeye salmon embryos incubated at 1.0°C had survival rates higher than 50%; chum and chinook salmon embryos exhibited 50% mortality at temperatures below 2.5 and 3.0°C, respectively; and even and odd-year pink salmon exhibited 50% mortality at 3.5 and 4.5°C, respectively (Beacham and Murray 1990). Conversely, 50% mortality occurred at temperatures above 13.5°C for coho salmon embryos, compared with 15-15.5°C for pink and sockeye salmon, and 16°C for chum and chinook salmon. The alevin stage is generally less temperature sensitive than the embryonic stages, with lower low-temperature thresholds, and higher high-temperature thresholds (Beacham and Murray 1990). Salmonid embryos and alevins can tolerate short periods during which temperatures are below or above incipient lethal levels (Bjornn and Reiser 1991).

Seymour (1956) carried out comprehensive studies on temperature effects on the development of chinook salmon from the egg to fingerling stage. Environmental temperature was correlated with the number of vertebrae, egg mortality, the number of abnormal fry, and the duration of the hatching period. For eggs reared at temperatures between 4.4 and 14.4°C, no differences were observed, but defects and mortality increased at both higher and lower temperatures. Combs (1965) identified lower (4.4–5.8"C) and upper (12.7- 14.2"C) temperature thresholds for normal development of sockeye salmon eggs.

Dissolved Oxygen. Embryos and alevins need high levels of oxygen to survive (Shirazi and Seim 1981). Field studies have demonstrated positive correlations between DO and survival for steelhead trout (Coble 1961) and coho salmon (Phillips and Campbell 1961). Phillips and Campbell (1961) suggested that DO levels must average greater than 8.0 mg/L for embryos and alevins to survive well.

In addition to being directly lethal, low levels of dissolved oxygen can have sublethal affects on salmonids as well. The rate of embryological development, the time to hatching, and size of emerging fry are all affected by low levels of dissolved oxygen. Garside (1966) found that the rate of embryonic development was increasingly retarded by progressively lower levels of dissolved oxygen (DO), resulting in delayed hatching. Doudoroff and Warren (1965) reported that DO levels below saturation resulted in increases in time to hatching and completion of yolk-sac absorption, as well as decreases in the size of alevins. Silver et al. (1963) and Shumway et al. (1964) observed that steelhead trout, coho salmon, and chinook salmon reared in water with low or intermediate oxygen concentration were smaller in size and had a longer incubation period than those raised at high DO. Similarly, Brannon (1965) found a positive relationship between DO and the size of sockeye salmon alevins at time of hatching. Alderdice et al. (1958) found that very low oxygen levels at early egg incubation stages produced severe morphometric abnormalities in chum salmon in addition to delaying hatching. Low DO levels stimulated eggs in an advanced stage of development to hatch prematurely, causing mortality.

Bjornn and Reiser (1991), summarizing four different studies, concluded that critical dissolved oxygen levels needed to meet respiratory demands vary with state of development. Early embryological states (pre-eyed) require the lowest levels of oxygen, while embryos nearing hatching have the highest DO requirements.

Turbidify and Sedimentation. Salmonids require gravels that have low concentrations of fine sediments and organic material for successful spawning and incubation. Bedload or suspended organic and inorganic materials that settle out over spawning redds affect the intragravel environment of salmonid embryos in several ways. Inorganic sediments, as discussed above, may clog substrate interstices and thereby diminish intragravel flows. In addition fine sediments may act as a physical barrier to fry emergence (Cooper 1959, 1965; Wickett 1958; McNeil and Ahnell 1964; Koski 1972; Everest et al. 1987). Eggs deposited in small gravel or gravel with a high percentage of fine sediments have lower survival to emergence (Harrison 1923; Hobbs 1937; Shapovalov and Berrian 1940; Shaw and Maga 1943; Koski 1966). McHenry et al. (1994) found that excessive fines (> 13% of sediments < 0.85 mm) resulted in intragravel mortality for coho salmon and steelhead trout embryos because of oxygen stress. Organic materials that enter the substrate interstices use up oxygen as they decompose (Bjornn and Reiser 1991), further reducing DO concentrations. In addition, salmon and trout avoid areas with high percentages of sand, silt, and clay (Burner 1951; Stuart 1953).

5.2.3 Rearing Habitat: Juveniles and Adult Residents

The abundance of juveniles and resident adult salmonids is influenced by the quantity and quality of suitable habitat, food availability, and interactions with other species, including predators and competitors (Bjornn and Reiser 1991). **As** noted in Section 4.2, the types of rivers and streams used for spawning and rearing differ among species and life stages. in addition, within the same system, sympatric populations of salmonids may segregate by channel unit types (e.g., pools, glides, riffles, cascades, off-channel areas) or by microhabitats within channel-unit types. The selection of specific microhabitats likely reflects a balancing among various factors, including the availability of food, the energetic costs of holding, risks of predation, and intra- and interspecific interactions. Bjornn and Reiser (1991) suggest that at any given time, certain environmental parameters may be better suited for some individuals, populations or species, while other parameters may not be as favorable yet must be kept in a suitable range for organism persistence. Consequently, there is no set of "optimal" habitat conditions for all species at all life stages.

Physical Structure

A variety of lentic and lotic habitats are potentially available to juvenile salmonids because of species-specific differences in ecological specialization. Comparison of habitat requirements among species is difficult: habitat selection is influenced by life stage, time of year, food availability, year-to-year variation in environmental conditions (e.g., flow, depth, temperature, food), and presence of other salmonids (Everest et al. 1985; Bjornn and Reiser 1991). Nevertheless, some species-specific affinities for habitats have been documented in the literature.

Shortly following emergence, the fry of many salmonids occupy shallow habitats along the margins of streams, moving into deeper and faster waters as they increase in size. Pink and chum salmon generally migrate to sea immediately after emergence. Sockeye salmon primarily use lakes as nursery areas, but will occasionally overwinter in sloughs, side channels, and spring areas (Burgner 1991). Within lakes, fry often use littoral areas for a month or more before moving offshore (Burgner 1991); riparian vegetation and woody debris may provide cover during this phase. Juvenile coho salmon tend to prefer pool habitats in summer and often move into side channels, sloughs, or beaver ponds for winter (Meehan and Bjornn 1991; Tschaplinski and Hartman 1983). Cutthroat trout also exhibit preference for pool habitats in summer, but the presence of other species such as coho salmon may cause trout to move into riffle habitats (Glova 1986). Steelhead trout typically prefer riffle habitats during summer (Everest et al. 1985) but may shift to pool habitats in winter or when coho salmon are not present (Bjornn and Reiser 1991). Juvenile chinook salmon are typically found in glide and riffle habitats with faster waters than typically used by coho salmon, though chinook do use pool habitats when available. Backwaters and side-channels that developed along unconstrained reaches in alluvial floodplains were historically important rearing habitats for many salmonid juveniles (Sedell and Luchessa 1982), and where these habitats remain intact they often contribute a disproportionate share of total salmonid abundance.

As detailed in Section 3.9.5, large woody debris interacts with natural channel-forming features such as boulders or bedrock to create different types of pool habitats (e.g., plunge pools, scour pools, eddy pools) and to increase hydraulic heterogeneity. The influence of large wood on both the formation of channel units and specific microhabitats creates habitat complexity that allows multiple species to coexist as an assemblage. in addition, large wood and associated pool habitats provide cover from predators and refuge habitats during storm events (Everest et al. 1985). Undercut banks and overhanging vegetation also serve as cover for juvenile anadromous and resident adult salmonids.

Substrate may also play an important role in habitat selection. in the summer months, boulders may provide both visual isolation from other fish and cover from predators. In winter, several salmonid species (e.g., steelhead, resident rainbow, and cutthroat trout as well as chinook salmon) have been observed to seek refuge in substrate interstices at low water temperatures (Chapman and Bjornn 1969; Bustard and Narver 1975; Campbell and Neuner 1985; Hillman et al. 1989). Gravel, cobble, and boulder substrates provide greater interstitial refugia than substrates dominated by sand or silt.

Flow and Depth

The amount of physical space available to juvenile and adult salmonids rearing in streams and the quality of that habitat is directly related to stream discharge (Everest et al. 1985). Within stream environments, salmonids select specific microhabitats where water depth and velocity fall within a specific range or where certain hydraulic properties occur (Table 5-5). These preferences in depth and velocity change both with season and life stage. Consequently, streamflow must be adequate to both satisfy minimum requirements for survival during periods of stress (e.g., low flow) as well as to provide specific microhabitat characteristics that are favorable to salmonid populations throughout their period of freshwater residence.

For many salmonids, smaller-sized fish tend to select shallower, slower moving waters than larger individuals (Chapman and Bjornn 1969; Everest and Chapman 1972; Moyle and Baltz 1985). Newly emerged fry may be vulnerable to downstream displacement by flow and typically select velocities lower than 10 cm/s (Bjornn and Reiser 1991). During summer months, salmonids often select

Table 5-5. Stream depths and velocities at holding sites of salmonids by age or size. From Bjornn and Reiser (1991). Reproduced with permission of the publisher.

Species and Source	Age* or Size	Depth (cm)	Velocity (cm/s)
Steelhead trout			
Bugert (1985)	31 - 44 mm	24	40
Everest and Chapman (1972)	0	< 15	< 15
	1	60 - 75	15 - 30
Hanson (1977)	1	51 mean	10 mean
	2	58 mean	15 mean
	3	60 mean	15 mean
Moyle and Baltz (1985)	0	35	7.3
	Juvenile	63	19.4
	Adult	82	28.6
Sheppard and Johnson (1985)	37 mm	< 30	< 25
Smith and Li (1983)	25 mm		4
	50 mm		8
	75 mm		18
	100 mm		24
	150 mm		24
Stuehrenberg (1975)	0	< 30	14 (range, 3 - 26)
	1	> 15	16 (range, 5 - 37)
Thompson (1972)	0	18 - 67	6 - 49
Chinook salmon			
Everest and Chapman (1972)	0	15 - 30	< 15
Konopacky (1984)	77 - 89 mm	55 - 60	12 - 30
			18 (dawn)
			12 (midday)
			25 (dusk)
Stuehrenberg (1975)	0	< 61	9 (range, 0 - 21)
	1	< 61	17 (range, 5 - 38)
Thompson (1972)	0	30 - 122	6 - 24
Steward and Bjornn (1987)	78 - 81 mm	40 - 58	8 - 10
Coho salmon			
Bugert (1985)	40 - 50 mm	24	39 (flume)
	0		15
	1		18
Nickelson and Reisenbichler (1977)	0	> 30	> 30
Pearson et al. (1970)	0		9 - 21
Sheppard and Johnson (1985)	62 mm	30 - 70	< 30
Thompson (1972)	0	30 - 122	5 - 24
Cutthroat trout			
Hanson (1977)	1	51 mean	10 mean
	2	56 mean	14 mean
	3	57 mean	20 mean
	4	54 mean	14 mean
Pratt (1984)	< 100 mm	32	10
	> 100 mm	62	22
Thompson (1972)	0, 1	40 - 122	6 - 49
Bull trout			
Pratt (1984)	< 100 mm	33	9
	> 100 mm	45	12

* Ages are in years or life stages, without units.

holding positions at moderate velocities but immediately adjacent to faster waters (Chapman and Bjornn 1969; Jenkins 1969; Everest and Chapman 1972). These positions are believed to confer the greatest energetic advantage to the fish. The amount of food delivered to a particular location is proportional to water velocity (Wankowski and Thorpe 1979; Smith and Li 1983). Consequently, fish that hold in water adjacent to faster feeding lanes can maximize food intake while minimizing energy expenditures associated with maintaining position in the current (Smith and Li 1983; Fausch 1984).

During winter months, metabolic demands and, thus, food requirements decrease as temperatures drop. Swimming ability also decreases with decreasing temperature (Brett 1971; Dickson and Kramer 1971; Griffiths and Alderdice 1972), and fish may be less able to maintain positions in fast waters for extended periods of time. **As** a result, salmonids tend to select slower water velocities, move to off-channel habitats, or seek refuge in substrate interstices when temperatures drop below a certain threshold temperature (Bustard and Narver 1975; Tschaplinski and Hartman 1983; Campbell and Neuner 1985; Johnson and Kucera 1985; Sheppard and Johnson 1985). Larger resident trout may abandon feeding sites in riffles and runs and move to slower-velocity pool habitats if substrate refugia are unavailable (Spence 1989).

For resident salmonids and juveniles of anadromous species that spend **a** year or more in freshwater, streamflow during the summer low-flow period must be adequate to prevent streams becoming excessively warm or drying up altogether. Under drought conditions, streams may become intermittent, and fish may be restricted to isolated pools. Such conditions can result in increased competition for food, reduced dissolved oxygen levels, increased physiological stress, and vulnerability to predators. Deep pools with groundwater inputs provide the necessary cover and thermal refugia.

Water Quality

Temperature. Juvenile and resident salmonids are variable in their temperature requirements, though most species are at risk when temperatures exceed 23–25°C (Bjornn and Reiser 1991). Upper and lower lethal temperatures as well as the "preferred" temperature ranges of several western salmonids is shown in Table 5-6. These values provide a general range of tolerable temperatures; however, the ability of fish to tolerate temperature extremes depends on their recent thermal history. Fish acclimated to low temperatures, for example, have lower temperature thresholds than those acclimated to warmer temperatures.

Table 5-6. Lower lethal, upper lethal, and preferred temperatures for selected salmonids. Based on techniques to determine incipient lethal temperatures (ILT) and critical thermal maxima (CTM). From Bjornn and Reiser (1991). Reproduced with permission of the publisher.

	Lethal temperature (°C)				
Species	Lower lethal*	Upper lethal†	Preferred temperature (°C)	Technique	Source
Chinook salmon	0.8	26.2	12–14	ILT	Brett (1952)
Coho salmon	1.7	26.0 28.8‡	12–14	ILT CTM	Brett (1952) Becker and Genoway (1979)
Sockeye salmon	3.1	25.8	12–14	ILT	Brett (1952)
Chum salmon	0.5	25.4	12–14	ILT	Brett (1952)
Steelhead trout	0.0	23.9	10–13		Bell (1986)
Rainbow trout		29.4 25.0		CTM ILT	Lee & Rinne (1980) Charlon et al. (1970)
Cutthroat trout	0.6	22.8			Bell (1986)

* Acclimation temperature was 10°C; no mortality occurred in 5,500 min.

† Acclimation temperature was 20°C unless noted otherwise; 50% mortality occurred in 1,000 min

‡ Acclimation temperature was 15°C.

Temperatures exceeding the upper "incipient lethal level" may be tolerated for brief periods, particularly during diel fluctuations, or may be avoided by seeking coldwater refugia provided by seeps or springs. Bull trout (not shown in table) appear particularly sensitive to warm waters. Temperatures higher than 14°C may act as a thermal barrier to migration of bull trout (OWRRI 1995). McPhail and Murray (1979) found that bull trout grew most rapidly at temperatures of 4°C, about 10°C colder than optimal growth temperatures for most species of *Oncorhynchus.* Lower lethal temperatures are near 0°C for most species of salmonids.

Many salmonid-bearing streams in the Pacific Northwest, particularly those in the southern, eastern, and low-elevation portions of the range, now experience maximum temperatures in summer that approach or exceed upper lethal levels for salmonids. Coldwater refugia in the form of springs, seeps, cold tributaries, and thermally stratified pools allow populations to persist in these streams that would otherwise be inhospitable. Nielsen et al. (1994) found that juvenile steelhead moved into thermally stratified pools when mainstem temperatures were between 23–28°C in a coastal northern California stream. Similarly, Li et al. (1991) reported that resident rainbow trout in an eastern Oregon stream selected natural and artificially created coldwater seep habitats when main-channel temperatures exceeded 24°C but showed no preference for coldwater areas when temperatures in the main channel dropped below 20°C.

Dissolved Oxygen. Salmonids are strong, active swimmers and require highly oxygenated waters. Maximum sustained swimming performance dropped off for coho and chinook salmon when DO concentrations decreased much below air-saturation levels (8–9 mg/L at 20°C) (Davis et al. 1963; Dahlberg et al. 1968). Alabaster et al. (1979) concluded that growth rate and food-conversion efficiency were probably limited if DO concentrations fell below 5 mg/L for Atlantic salmon. Davis (1975) estimated that salmonids would suffer no impairment if DO concentrations reniained near 8 mg/L and determined that DO deprivation would begin at approximately 6 mg/L. High water temperatures, which decrease oxygen solubility, further increase the stress on fish caused by low DO concentrations. A recent literature review resulted in criteria for salmonids presented in Table 5-7 (ODEQ 1995). The dissolved oxygen criteria developed for Idaho, Oregon, and Washington provide greater protection to salmonids than EPA's national minima.

Turbidity. Turbidity is elevated in all streams for short durations during storm and snowmelt events. Juveniles and adults appear to be little affected (Sorenson et al. 1977) by these transitory episodes, though Bisson and Bilby (1982) reported that coho salmon avoided water exceeding 70 nephelometric turbidity units (NTU), levels that may occur in some watersheds with high erosion potential. In a laboratory setting, juvenile coho salmon and steelhead trout exhibited reduced growth rates and higher emigration rates in turbid streams (25–50 NTU) compared to clear streams (Sigler et al. 1984). Lloyd et al. (1987) found that juvenile salmonids avoided chronically turbid streams including glacially influenced streams and those disturbed by human activities, Turbidity also influences foraging behavior of juvenile anadromous and adult resident salmonids by reducing the distance from which they can locate drifting prey.

5.2.4 Juvenile Migration

Depending upon the species or population, some juvenile salmonids migrate to the sea or to lakes, while others remain in a relatively small reach of stream for their entire lives (Everest et al. 1985; Bjornn and Reiser 1991). All species require unobstructed (either physically or chemically) access to upstream or downstream reaches for migration or dispersal to feeding grounds. In addition, species and stocks differ in their migratory behavior (i.e., timing and speed). For example, some species (e.g., pink and chum salmon) may move rapidly to the ocean over a few hours or days, while others (e.g., chinook salmon) may gradually move downstream over several weeks or months. These different behaviors entail substantially different habitat requirements during the migration period.

Physical Structure

Migrating fish are particularly vulnerable to predation because they often are concentrated and may move through areas with limited cover and high abundances of predators (Larsson 1985). Physical structure in the form of undercut banks and large woody debris provides refugia during resting periods and cover from predators. Juveniles that migrate to lakes, such as sockeye salmon or adfluvial resident populations, may be traveling upstream or downstream. In addition to cover from predators, these fish may require holding and feeding stations during their migrations. Artificial obstructions such as dams and diversions may impede migrations where they create unnatural hydraulic configurations.

Flow and Depth

Streamflow is important in facilitating downstream movement of salmonid smolts. Smolt migration is believed to be regulated by "priming" factors, such as photoperiod and temperature, that alter the

Table 5-7. Guidance for relating dissolved oxygen criteria to use protection. From ODEQ (1995).

	Concentration (mg/L)				
Class	30-day mean	7-day mean	7-day minimum	Minimum	Use/level of protection
Salmonid spawning		11		9 6*	Salmonid spawning and incubation of embryos. Low risk of impairment to aquatic community of salmonids, other native fish, and invertebrates.
Coldwater	8		6.5	6	Principally coldwater communities, salmon, trout, invertebrates, other native coolwater species througout all or most of the year. Juvenile anadromous salmonids may rear throughout the year. Low level risk of impairment for these groups.
Coolwater	6.5		5	4	Mixed native coolwater species, such as sculpins, and coolwater aquatic life. Provides migratory route for salmon and trout. Salmonids and other biota may be present during part or all of the year but may not dominate community structure. Slight level of risk to community.
Warmwater	5.5			4	Native warmwater fish; non-native species, salmonid migration; waterbodies may not naturally support native coolwater communities.
No risk	No change from natural				The only criteria that provides no additional risk to the resource is no change from background.

* Intragravel dissolved oxygen.

disposition of the fish in anticipation of downstream migration and "releasing" factors, including changes in temperature or streamflow, that trigger movement once a state of physiological "readiness" is obtained (Groot 1982). Dorn (1989) found that increases in streamflow triggered downstream movement of coho salmon in a western Washington stream. Similarly, Spence (1995) also found short-term increases in streamflow to be an important stimulus for smolt migration in four populations of coho salmon. Thus the normal range of streamflows may be required to maintain normal temporal patterns of migration.

Streamflow is also important in determining the rate at which smolts move downstream, although factors influencing the speed of migration remain poorly understood. Bjornn and Reiser (1991) state that the time required to travel from the Salmon River in Idaho to the Dalles Dam increased by as much as 30 days during low-flow periods following the construction of six dams on the Columbia-Snake system. In other systems, the migration speed of individuals may not be correlated to streamflow. This may occur in part because of the changing physiological disposition of fish during the run period with later migrants undergoing a more rapid smolt transformation as water warms.

Water Quality

Temperature. Temperature affects migration timing of smolts in two fundamental ways: by influencing the rate of growth and physiological development and by affecting the responsiveness of fish to other environmental stimuli (Groot 1982). Consequently, alteration of thermal regimes through land-use practices and dam operations can influence the timing of migration. Holtby (1988) found that coho salmon smolts emigrated approximately 8 days earlier in response to logging-induced increases in stream temperatures. In addition, the age-class distribution was shifted from populations evenly split between one- and two-year old smolts to populations dominated by one-year old fish. A single year of poor ocean conditions will have a greater effect on a particular year class if the majority of smolts migrate at the same age, rather than spreading the risk over two years.

The specific temperature requirements of juvenile anadromous salmonids during their seaward migration are not well documented. Sockeye smolts have been reported to migrate at temperatures ranging from 2–10°C (Burgner 1991). Coho salmon have been observed to migrate at temperatures as low as 2.5°C and as high as 13.3°C (Sandercock 1991); however, most fish migrate before temperatures reach 11–12°C. Ocean-type chinook typically migrate during March and April at temperatures between 4.5°C and 15.5°C (Healey 1991), whereas stream-type chinook smolts tend to migrate 1 to *2* months later when conditions are substantially warmer. Once temperatures exceed a threshold level in the spring, salmon smolts will revert to a presmolt physiology and remain within the stream.

Dissolved Gases. Supersaturation of dissolved gasses (particularly nitrogen) has been found to cause gas bubble disease in upstream and downstream migrating salmonids (Ebel and Raymond 1976). Steelhead trout appear to be more susceptible than salmon to the disease because salmon have been found to better sense and avoid highly supersaturated waters (Stevens et al. 1980). However, all salmonids are susceptible. Although we found no information regarding dissolved oxygen requirements during seaward migration, it is likely that DO near saturation levels is required during this physiologically stressful period.

Turbidity. Turbid waters have been mentioned as affecting migration but little documentation is available in the literature. Thomas (1975) found fry migration increased as turbidity increased. Lloyd et al. (1987) found that turbid streams were avoided by juveniles except when the fish must pass through them along migration routes. There is also some evidence that diel migrations of salmonids is influenced by turbidity. Many salmonids tend to migrate during the evening hours (Burgner 1991), presumably to avoid predation. However, in streams with higher turbidity, migrations may be evenly dispersed during both the day and night.

6 Effects of Human Activities

Land-use practices—forestry, grazing, agriculture, urbanization, and mining—disrupt aquatic ecosystems by altering watershed processes that ultimately influence the attributes of streams, lakes, and estuaries. In this section, we review specific mechanisms by which human activities directly or indirectly affect aquatic ecosystems. With the exception of chemical contamination, most effects on watershed processes result from changes in vegetation and soil characteristics, which in turn affect the rate of delivery of water, sediments, nutrients, and other dissolved materials from uplands to stream channels. Within the riparian zone, land-use activities can alter the amount of solar radiation reaching the stream surface, affect the delivery of coarse and fine organic materials to streams, and modify fluvial processes that affect bank and channel stability, sediment transport, seasonal streamflow patterns, and flood dynamics. Disconnecting streams from their floodplains further alters hydrologic processes, nutrient dynamics, and vegetation characteristics.

Dams and water diversions affect salmonids by blocking or hindering migrations of fish, by altering the physical (e.g., temperature, flow, sediment routing) and chemical characteristics of streams, and by causing changes in stream biota. Other activities that influence salmonids and their habitats include wetland removal, harvesting of salmon, introduction of non-native species and hatchery salmonids, eradication of beaver, and activities associated with river, estuarine, and ocean traffic.

6.1 Forestry

Forest vegetation covers approximately 46% of the combined land surface of Washington, Oregon, and Idaho, including 34% of nonfederal lands and 58% of Federal lands (Pease 1993)'. Most commercial harvesting of timber is for softwoods, primarily Douglas-fir, ponderosa pine, hemlock, Sitka spruce, and lodgepole pine. Industry-owned lands, despite constituting only 18% of the softwood growing stock, accounted for 44% of the total softwood harvest in the region in 1986. In contrast, national and State forests constitute 70% of the growing stock, but provided only 46% of the softwood harvest (Jensen 1993). Intense production from nonfederal lands is likely to continue or increase as Federal timber supplies diminish.

Logging in the Pacific Northwest began in the mid 1800s, and by the 1860s the timber industry was well established. By 1880, forests along Puget Sound, as well as many rivers and streams, had been cleared for three or more kilometers inland (Sedell and Luchessa 1982). Throughout the 1900s forest harvest has continued, and the effects of logging have become pervasive across the region. Early forest practices were particularly damaging to stream environments. Splash damming was commonly used to float logs down to the sawmills, a practice that has had long-lasting effects on channel morphology and the abundance of large woody debris. Clear-cuts often included riparian forests, which yielded large quantities of wood that were easily transported downstream. Debris jams were routinely removed at the behest of biologists, who believed they hindered migration of anadromous fishes. Today the functional importance of large woody debris to salmonids is well documented (Bisson et al. 1987; Hicks et al. 1991a; Naiman et al. 1992) and State forest practice rules have been modified to reflect this knowledge. But despite recent improvements in forest practices, a legacy of past practices and cumulative effects will hamper our ability to quickly reverse habitat changes accrued from logging practices.

In the sections that follow, we review the effects of forest practices on watershed processes and salmonid habitats. We use the term "forest practices" to include all activities associated with the access, removal, and re-establishment of forest vegetation, including road construction, timber harvest, site preparation, planting, and intermediate treatments. Understanding the effects of these practices on natural processes will foster improved management, providing greater protection to salmonid habitats.

6.1.1 Effects on Vegetation

Forest practices directly influence vegetation within a watershed through the removal of trees during harvest, thinning, and road construction, and

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'Percentages listed in Table 7 of Pease (1993) for nonfederal lands are in error. The correct total area for nonfederal lands is 33,616,655 hectares (83,066,500 acres). (P. Jackson, Geosciences, Oregon State University, personal communication.)

through manipulations of understory and ground vegetation designed to increase the vigor of desired species and inhibit growth of understory vegetation (e.g., burning or mechanical and chemical treatments). In addition, forest vegetation is indirectly affected by changes in site conditions following harvest. Removal of overstory vegetation can change local microclimate, soil moisture and stability, ground cover, and susceptibility to erosion, all of which influence the re-establishment of vegetation in the harvested area (Beschta et al. 1995). Soil compaction by ground-based equipment can reduce infiltration of water, thereby hindering the re-establishment of seedlings or the growth of established vegetation. In addition, as tree roots die after logging, subsurface spaces (macropores) become compacted or filled with sediment, reducing infiltration of water and reducing aeration in the soils. When porosity is reduced below 20%–25%, root growth is retarded. Mixing of mineral and organic soil layers also strongly influences the revegetation process (Beschta et al. 1995).

The magnitude of vegetation change and the succession of vegetation following logging depend on the type and degree of disturbance. For highly disturbed sites, early succession is dominated by colonizing annual and herbaceous species, followed by dominance/codominance of perennial species, and finally by dominance of overstory species. Where disturbance is less severe, residual species may dominate the early successional stages. Beschta et al. (1995) provide a more thorough review of the effects of forest practices on regeneration of vegetation.

At the landscape level, forest practices have resulted in substantial modification of species and age composition of western forests. Natural forests typically exhibit a mosaic of patches in different states of ecological succession. These mixed-age, multi-species plant assemblages have been replaced with even-aged forest plantations dominated by a single species. Riparian forests have been especially affected in areas where rapid growth of hardwood species (e.g., alder and maple) and shrubs (e.g., salmonberry) has precluded re-establishment of coniferous species (Bisson et al. 1987). In coastal streams, riparian areas outside of wilderness areas are dominated by alder and big leaf maple (FEMAT 1993). Certain conifers, such as western hemlock and Sitka spruce, frequently regenerate on partially decomposed nurse logs that are elevated above the forest floor (reviewed in Harmon et al. 1986). Removal of downed trees from the riparian zone may affect re-establishment of these species.

6.1.2 Effects on Soils

Forest practices can result in significant disturbance to soils, including increased compaction, scarification, and mixing of soil layers. The degree and effects of compaction are influenced by a number of factors, including the total area compacted, the soil type and moisture content, the equipment used, and the number of passes the vehicle makes over the site. Cafferata (1992 in Beschta et al. 1995) reviewed a number of studies and determined that 10%–40% of a harvest area may be compacted during tractor logging.

The effects of soil compaction appear to be of long duration. Studies have estimated recovery times from 10–50 years, with estimates as long as 90–110 years in an arid high-elevation site (Webb et al. 1986; Cafferata 1992). Duration of compaction depends upon depth of compaction, soil texture, soil temperature and moisture regimes, and biological activity. Recovery time increases with increasing depth of compaction. Soil recovery occurs more rapidly in clay soils that shrink and swell with changing moisture content, high elevation soils that are subjected to freezing and thawing, soils with high organic content that cushions them from compaction, and soils with high biological activity (e.g., burrowing rodents, earthworms, insects, soil microbes) (Beschta et al. 1995).

6.1.3 Effects on Hydrology

Timber harvest and its associated road construction and site preparation practices can have significant effects on hydrologic processes that determine streamflow. In most cases, the removal of vegetation increases the amount of water that infiltrates the soil and ultimately reaches the stream by reducing water losses from evapotranspiration. However, in forested systems where fog drip contributes significantly to total precipitation (Harr 1982), harvesting trees may have little effect on the total amount of water reaching the stream. Soil compaction can decrease infiltration and increase the likelihood of surface runoff. Roads can affect the routing of water by intercepting subsurface flow and diverting it down drainage ditches, effectively increasing drainage density within a watershed (Sidle et al. 1985). King and Tennyson (1984) observed altered hydrology when roads constituted 4% or more of catchment area (i.e., 4% was "roaded"). In snow-dominated systems, logging can influence the spatial distribution of snow on the ground, as well as the energy transfer processes that affect the melting rate of snowpack (Chamberlin et al. 1991). The effect of logging on hydrologic processes can change annual water yield, the magnitude and timing of peak flows, and the magnitude of summer low flows. The effects of logging on hydrologic processes are reviewed in two recent syntheses (Chamberlin et al. 1991; Beschta et al. 1995), and the material presented below is based primarily on these analyses.

Water Yield

In most instances, clear-cutting has been found to increase total water yield. In western Oregon, increases in water yield in the first 1–5 years following logging have ranged from a few centimeters to almost 65 cm (25 inches) (Beschta et al. 1995). The largest increases in yield occur in areas of high precipitation and high evapotranspiration. In these areas, increased yield resulted primarily from reduced evapotranspiration losses. One case study in the Bull Run watershed of Oregon found that water yields decreased slightly after logging (Harr 1982). Apparently, the decrease in evapotranspiration losses was offset by a decrease in the amount of fog-drip that normally occurred in the forested watershed.

In forests east of the Cascade Range, increases in water yield after logging are not as great. In a snow-dominated system in the Blue Mountains of Oregon, no increase in water yield was observed following logging (Fowler et al. 1987). The authors suggested that higher wind speeds after clear-cutting increase evaporation from snowpack, offsetting reductions in transpiration. Elsewhere in eastern Oregon and Washington, forested watersheds are likely to experience minor changes in total water yield. Beschta et al. (1995) speculated that in areas with low precipitation (< 30–51 cm [< 15–20 inches]) increases in water yield are likely to be negligible. Water yields in interior ponderosa pine and Douglas-fir forests are generally low to begin with because of high evaporation demands. Consequently, reductions in transpiration losses may be compensated by higher evaporation losses. In areas with higher precipitation, increases in water yield may be somewhat greater but still less than those observed in westside systems (Ziemer 1986; Beschta et al. 1995). Small increases in yield from snow-dominated systems in British Columbia and Colorado have also been reported (Hibbert 1967; Cheng 1989).

In addition to being affected by forest vegetation and climatic conditions, increases in water yield also depend on the percentage of the land area that is harvested or roaded (Hewlett and Nutter 1970; Trimble and Weirich 1987). In general, the increase in yield is directly proportional to the size of the area logged. However, for patch cuts, removal of vegetation may result in a smaller increased yield than predicted by area alone because of increased utilization of available moisture by vegetation in surrounding uncut areas (Beschta et al. 1995). For the same reason, selective harvesting or thinning may have minimal effect on water yield (Hibbert 1967). Bosch and Hewlett (1982) reviewed over 90 watershed studies and concluded that increased yield usually occurs after 20%–30% of a watershed has been harvested.

The effects of logging on total water yield persist until the transpiration demands of recovering vegetation approach those of uncut forests. In forests west of the Cascade Crest, return to natural conditions may take 30–40 years if no further disturbances occur in the watershed (Harr and Cundy 1992; Stednick and Kern 1992). Brush removal by mechanical means, chemical treatments, or burning (to aid re-establishing desired trees) can slow the rate of recovery. However, in general, these activities are practiced only until seedlings attain sufficient height to shade out competing species. Thus effects of these practices on water yield are likely to be short term.

Timing of Runoff

In rain-dominated systems, the largest increases in water yield because of logging generally occur during periods where both precipitation and transpiration rates of vegetation are relatively high, usually the fall (Chamberlin et al. 1991) and spring (Beschta et al. 1995). With reduced transpiration, soil moisture is rapidly replenished with the onset of rains in the fall and subsurface flow to stream channels commences (Rothacher 1971; Harr et al. 1979). Evapotranspiration losses from mature forests are comparatively small during the winter because of low temperatures and high humidity and, consequently, increased yield in winter is generally smaller (Chamberlin et al. 1991); however, in the spring, the differences between transpiration losses in mature forests and those in clear-cuts are again greater, and increases in water yield may be higher than in winter (Beschta et al. 1995).

In snow-dominated systems, increases in water yield generally occur during the early spring snowmelt period. The loss of shading following removal of the forest canopy can accelerate snowmelt, resulting in an earlier peak in the stream hydrograph. In snow-dominated systems, solar radiation is the primary factor influencing the rate of snowmelt (Chamberlin et al. 1991). In the snow-rain transition zone of the western Cascades snowmelt is driven primarily by convective transfer of sensible and latent heat to the snowpack (Harr 1986). Opening up the forest canopy can increase wind speed and turbulence, facilitating more rapid melting.

Peak Flows

A recent review (Beschta et al. 1995) of effects of timber harvest on peak flows in systems in the Pacific Northwest indicates a high degree of variability among sites. In rain-dominated systems of the Coast Range, most studies have indicated increases in peak flows following logging, particularly those occurring in fall (Table 6-1). In a few cases, increases have been insignificant, and in one case, a decrease in peak flows was observed.

Where increases in peak flow occur, they likely result from reduced evapotranspiration losses with removal of the forest canopy and more rapid routing of water to the stream channel because of soil compaction and roads.

In transient-snow systems of western Oregon responses of peak flows are similarly variable (Table 6-2). Several studies have indicated increases of 10%–200% in peak flows (Rothacher 1973; Harr et al. 1979), while others have shown no change or decreases (Harr et al. 1979; Harr et al. 1982; Harr and McCorison 1979). Harr (1986) re-analyzed published data and found that studies showing decreases in peak flows were inconclusive. In systems where harvest has increased peak flows in the transient-snow zone, it is believed that vegetation removal increased delivery of water to the soil from the snowpack during rain-on-snow events (Harr 1986). Coffin and Harr (1992) used lysimeters placed under the snowpack to confirm increased melt rates and delivery of water to the soil during rain-on-snow events. Maximum differences in melt rates between open and forested plots occurred when rain events were accompanied by relatively high temperatures and wind speeds, apparently because of increased transfer of sensible and latent heat to the snow. Coffin and Harr (1992) report that effects of harvesting on peak flows were still evident in 20–25 year-old plantations.

In snow-dominated systems, peak flows have generally shown little or no change following logging (Table 6-2) although studies are limited in geographic

Table 6-1. Effects of timber harvesting on peakflows in coastal areas of the Pacific Northwest. Modified from Beschta et al. (1995). Reproduced with permission from the principal author.

Watershedl Location	Size (acres)	Geology/Soils	Harvest method and percent	Peakflow effect	Reference
OREGON					
Needle Branch (Alsea watershed)	173	sandstone shallow GL-GCL[*]	cc[†] 82%	fall: +50% winter: +19%	Hsieh (1970) Harris (1977) Harr et al. (1975)
Deer Creek–Main (Alsea watershed)	748	sandstone shallow GL-GCL	cc 26%	all: ns[‡]	Hsieh (1970) Harris (1977) Harr et al. (1975)
Deer Creek–2 (Alsea watershed)	138	sandstone shallow GL-GCL	cc 90%	fall: +51% winter: +20%	Hsieh (1970) Harris (1977) Harr et al. (1975)
Deer Creek–3 (Alsea watershed)	99	sandstone shallow GL-GCL	cc 65%	fall: +50% winter: +30%	Hsieh (1970) Harris (1977) Harr et al. (1975)
BRITISH COLUMBIA					
West Coast–I	57	quartz GSL[§]	cc 71%	all: –22%	Cheng et al. (1975)
Carnation Creek B	2,470	volcanic coarse soil	cc 41%	all: ns	Hetherington (1987)
Carnation Creek	30	volcanic coarse soil	cc 90%	all: +20%	Hetherington (1987)
Jamieson Creek	739	na[¶]	cc 19.2%	winter: +13.5%	Golding (1987)
CALIFORNIA					
South Fork Caspar Creek	1,047	sandstone coarse soil	sc[#] 60%	small: +107% large: ns	Ziemer (1981)

[*] gravelly loam-gravelly clay loam (GL-GCL)
[†] clear-cut (cc)
[‡] not significant (ns)
[§] gravelly sandy loam (GSL)
[¶] not available (na)
[#] shelterwood (sc)

Table 6-2. Effects of timber harvesting on peakflows in interior areas of the Pacific Northwest. Modified from Beschta et al. (1995). Reproduced with permission of the principal author.

Watershed/ Location	Size (acres)	Geology/ soils	Harvest method and percent	Peakflow effects	Reference
RAIN-ON-SNOW					
Watershed—I* (OR)	235	basalt/GL[†]	cc* 100%	fall: +200% winter: ns[§]	Rothacher (1973)
Watershed—3 (OR)	249	basalt/GL	cc 25%	mean: +10%	Rothacher (1973)
Watershed—6 (OR)	32	basalt/GL	cc 100%	all: ns	Harr et al. (1982)
Watershed—7 (OR)	52	basalt/GL	sc[¶] 60%	all: ns	Harr et al. (1982)
Watershed—I0 (OR)	25	basalt/GL	cc 100%	ROS[#]: –36% rain: ns	Harr and McCorison (1979)
Coyote Creek—I** (OR)	170	basalt/ GL-GCL[††]	sc 50%	mean: +30% large: +48%	Harr et al. (1979)
Coyote Creek—2 (OR)	168	basalt/ GL-GCL	cc 30%	all: ns	Harr et al. (1979)
Coyote Creek—3 (OR)	121	basalt/ GL-GCL	cc 100%	mean: +44% large: +35%	Harr et al. (1979)
SNOWMELT					
High Ridge—I** (OR)	73	fractured basalt/SL[§§]	cc 43%	all: ns	Fowler et al. (1987)
High Ridge—2 (OR)	60	fractured basalt/SL	sc 50%	all: ns	Fowler et al. (1987)
High Ridge—4 (OR)	292	fractured basalt/SL	cc 22%	all: ns	Fowler et al. (1987)
Camp Creek (BC)▪▪	8,373	granite/ coarse soils	cc 30%	annual: –9 to +35%	Cheng (1989)

* Watersheds in H. J. Andrews Experimental Forest, western Cascades region
[†] gravelly loam (GL)
[‡] clear-cut (cc)
[§] not significant (ns)
[¶] sheltetwood (sc)
[#] rain-on-snow (ROS)
** Coyote Creek, South Umpqua Experimental Forest, Western Cascades Region
[††] gravelly loam—gravelly clay loam (GL-GCL)
[‡‡] High Ridge Evaluation Area, Blue Mountains Region
[§§] silt loam (SL)
[¶¶] Camp Creek, southern British Columbia, Canada

distribution. In most of these studies, the percentage of the watershed cut has been less than 50%. Despite the lack of conclusive data, it is reasonable to predict increased peak flows following logging. Snow accumulation is generally higher in open patches created by logging (Chamberlin et al. 1991), though it is unclear whether this is merely a redistribution of snow over the watershed or an actual increase in availability. Increased wind speeds in cleared areas may accelerate melting, leading to more rapid runoff and higher peak flows.

Low Flows

Increases in summer low flows have been observed following logging in a number of systems in the Pacific Northwest. Harr and Krygier (1972) documented average increases in summer flows of 60% following logging of a Coast Range stream in

Oregon. Somewhat larger increases were observed in a western Cascade stream (Rothacher 1970). Keppeler and Ziemer (1990) noted increases in summer flows in a northern California stream, but found that the increases disappeared within five years. Studies in drier, interior climates have been less conclusive. Cheng (1989) reported increases in summer streamflows that persisted for six years after logging of a basin in the interior of British Columbia. However, Troendle (1983) found no increase in summer low flows after logging in Colorado. Where increases in summer flows occur, they likely result from reductions in evapotranspiration losses.

Few long-term studies of effects of logging on low flows have been performed. A notable exception is Hicks et al. (1991b) who found that August streamflows in a central Oregon Cascade stream increased for 8 years following logging, but decreased for 18 of the next 19 years. On average, August streamflows were 25% lower than in prelogging years. The authors attributed reductions in streamflow to the replacement of coniferous vegetation with more water-consumptive hardwood species. Thus, the long-term effects of logging on streamflows likely depend on vegetation composition before and after harvest,

6.1.4 Effects on Sediment Transport

Forest practices can substantially increase delivery of sediments to streams through both surface erosion and mass wasting. The effect of forest practices on sediment transport depends on a number of local site conditions including climate, vegetation, topography, and soil type, as well as on specific aspects of the activity, including the type and areal extent of disturbance and the proximity of the disturbance to the stream channel. Thus, the relative effects of roading, timber harvest, site preparation, and other forest practices on sediment production vary with location (Beschta et al. 1995).

Furniss et al. (1991) concluded that forest roads contributed more sediment than all other forest activities combined on a per unit area basis. Summarizing results from nine different studies, they reported that mass wasting associated with roads produced 26–346 times the volume of sediment as undisturbed forests. Mass failures were attributed to poor road location, construction, and maintenance, as well as inadequate culverts. Beschta (1978) found that, in three out of eight years, suspended sediment increased significantly from two catchments in the Coast Range, primarily as a result of mass failures from roads. Mass failures associated with roads most commonly occur on cut and fill slopes, but may also begin where end-haul material is deposited on a hillslope (Dent **1993).** In addition, channel constrictions at road crossings may lead to bank sloughing and bank erosion.

Surface erosion from roading also constitutes a significant source of chronic sediment inputs (Beschta et al. 1995). Splash erosion mobilizes sediment on exposed road surfaces, and runoff from compacted surfaces may also facilitate sediment transport. Water diverted into ditches along roads gathers energy and can cause significant erosion at the outlets of cross-drain culverts (Beschta et al. 1995). Montgomery (1993), studying three small catchments, found that roads decreased the drainage area needed to support a channel head and thereby increased the length of the channel. He attributed this phenomenon to lower infiltration or greater runoff on roads. He also demonstrated that channel expansion was a function of catchment area. The combined effects of mass wasting and surface erosion can lead to elevated sediment levels in streams even when only a small percentage of a watershed is roaded. For example, Cederholm et al. (1981) reported increased sediments in salmonid spawning gravels when roads exceeded **3%** of the total basin area.

Sediment delivery from other forest activities, including harvest, yarding, and site preparation, may be increased via several mechanisms. Loss of the protective vegetative cover can increase splash erosion and decrease slope stability (Swanston et al. 1980; Marcus et al. 1990). Yarding activities cause extensive soil disturbance and compaction which may increase splash erosion and channelized runoff. Ground-based vehicles moving logs from felled trees and skidding logs to landing sites compact and scarify the soil. Compaction of the decomposing root systems reduces the infiltration capacity of these channels, leading to slumps, landslides, and surface erosion (Everest et al. **1987).** Loss of the humic layer through mechanical disturbance and fire further increases the potential for surface erosion.

The quantity of sediments delivered to the stream channel depends upon the integrity of the riparian zone, the intensity of disturbance, the areal extent of the disturbance, the proximity of the disturbance to the stream channel, and slope steepness. Site disturbance may be intensified by a hot bum following harvest which creates extensive areas of bare soil (Everest et al. 1987). Piling and burning versus broadcast burning will also intensify the site disturbance and increase sediment delivery rates. Riparian buffer strips and buffer strips below roads can trap sediments, significantly reducing the delivery rate (Swanston 1991).

The fate of sediments once in the channel also depends on the nature of sediments (coarse versus fine) and local site characteristics. Although surface erosion is less dramatic and less evident than mass wasting, it may be more detrimental to stream biota because the delivery of particles occurs over a longer time, and those particles are smaller and more likely to become embedded in coarser substrates. Bilby (1985) found that sediments from road crossings were flushed from a fourth-order stream reach of 2% gradient that drained a small (5.5 km^2) catchment with a relief ratio of 0.10. Presumably, these sediments (2–151 $mg \cdot L^{-1}$ above control levels) were

Table 6-3. Summary of summer temperature changes associated with management activities on forested watersheds in the Pacific Northwest. From Beschta et ai. (1987). Reproduced with permission from the publisher.

Location	Treatment	Stream temperature variables	Temperature change (°C)	Reference
Alaska (Southeast)	Clear-cut and natural openings	ATemperature per 100 m of channel	+0.1 – 1.1°C per 100 m; Avg = 0.7°C per 100 m	Meehan (1970)
British Columbia (Vancouver Island)	Logged (Tributary H)	Average Jun–Aug diurnal temperature range	+0.5 – 1.8°C over pretreatment levels	Holtby and Newcombe (1982)
	Logged and burned (Tributary J)	Average Jun–Aug diurnal temperature range	+0.7 – 3.2°C over pretreatment levels	Holtby and Newcombe (1982)
Oregon (Cascades)	Clear-cut	Average Jun–Aug maximum	+4.4 – 6.7°C	Levno and Rothacher (1967)
	Clear-cut and burning	Average Jun–Aug maximum	+6.7 – 7.8°C	Levno and Rothacher (1967)
Oregon (Coast Range)	Clear-cut	Average Jul–Sep maximum	+2.8 – 7.8°C	Brown and Krygier (1967)
	Clear-cut and burning	Average Jul–Aug maximum	+9 – 10°C	Brown and Krygier (1970)
Oregon (Cascades)	Mixed clear-cut and forested reaches	A Temperature per 100 m of channel	+0 – 0.7°C per 100 m	Brown et al. (1971)
	Tractor stripped area	A Temperature per 100 m of channel	+15.8°C per 100 m	Brown et al. (1971)

deposited downstream. In a separate study, Duncan et al. (1987) reported that first- or second-order channels with high amounts of wood debris retained 55% of road-crossing sediments at flows up to 7% of bankfull. Thus stream gradient and retentive in-channel structures appear important in determining whether sediments are deposited locally or transported downstream.

6.1.5 Effects on Thermal Energy Transfer and Stream Temperature

Logging most directly affects energy transfer by reducing shade provided by riparian vegetation, which increases the amount of direct solar radiation reaching the stream surface. The increase in energy reaching the stream depends on the amount of shading lost. Measurements from an old-growth Douglas fir forest in western Oregon indicated shading averaged 84% (Summers 1983). Brazier and Brown (1973) reported that angular canopy densities generally fall between 80% and 90% in old-growth stands in western Washington (cited in Beschta et al. 1987). In eastern Oregon, natural canopy density is somewhat less. Slightly lower shading (75%) has been reported for a stream in northern California (Erman et al. 1977), and Anderson et al. (1993) estimated shading in old-growth forests of the Upper Grande Ronde basin in eastern Oregon to be around 72%. Thus, the magnitude of increase in stream temperatures following canopy removal is likely to differ across the region (Table 6-3).

Removal **of** riparian canopy also **affects** other energy transfer processes including convection, evaporation, and advection. Convective and evaporative heat exchange are both affected by wind speed (see Section 3.7), which generally increases as riparian vegetation is removed. Consequently, convective exchange as well as evaporative losses tend to increase slightly following logging (Brown 1969). The removal of vegetation from upslope areas generally allows greater heating of the soil surface during the summer months. Rain falling in the early part of the rainy season may pick up additional heat as it passes through the soil and infiltrates subsurface aquifers, resulting in increases in ground water temperature after logging.

Removal of riparian canopy has been shown to have two major effects on temperatures of smaller streams in the Pacific Northwest: increased maximum temperatures (particularly in summer) and increased diel fluctuations (Beschta et al. 1987; Beschta et al. 1995). For coniferous forests of the Coast Range and western Cascades, increases in average summer maximum temperatures because of clear-cutting have ranged from about 3 to 8°C (Table 6-3); (Beschta et al. 1987). Increases up to 10°C have been observed when clear-cutting has been followed by slash burning (Brown and Krygier 1970). Increases in annual maximum daily temperatures can be substantially greater. Hall and Lantz (1969) reported increases in maximum daily stream temperatures of up to 16°C in years immediately following logging of a small watershed in the Coast Range of Oregon. Holtby (1988) reported that average monthly water temperatures increased from 0.7 to 3.2°C following logging of the Carnation Creek (British Columbia) watershed, with the largest increases occurring in May–September and the smallest increases in December and January. These changes persisted for at least seven years after logging. Average diel temperature fluctuations increased by as much as 3.7°C in two Carnation Creek tributaries that had diel fluctuations of less than 1°C prior to logging (Holtby and Newcombe 1982). Hall and Lantz (1969) reported that midsummer diel fluctuations of 15°C were common in Needle Branch, Oregon, after logging. Documentation of temperature changes resulting from logging east of the Cascade range is sparse. Because the degree of shading provided by more open forest types (e.g., ponderosa pine) is lower than for coastal and western Cascade streams, the increase in temperatures resulting from canopy removal might be expected to be slightly less. Nevertheless, because many streams east of the Cascades approach the maximum thermal tolerance level for salmonids during the summer, smaller increases in temperature might be equally or more detrimental to salmonids.

Although summer stream temperatures have been the focus of most research on the effects of logging on stream temperatures, changes in winter stream temperatures may also occur. Theoretically, the loss of riparian vegetation allows for greater radiative cooling at night during the winter months, potentially decreasing winter temperatures. However, Holtby (1988) reported increases in February–April mean temperatures of 1–2°C. Increases in groundwater temperatures following canopy removal may have been responsible for the increase in winter temperatures. Hall and Lantz (1969) also noted similar increases in temperatures during the winter in a coastal Oregon stream after the entire basin was clear-cut.

The magnitude of temperature change following removal of riparian vegetation depends on the size of stream and channel morphology. Because stream discharge and depth increase downstream, the ability of solar radiation to effect stream temperatures also diminishes with increasing stream size (Beschta et al. 1995). Moreover, the amount of shading provided by riparian vegetation decreases as streams become larger and wider. Consequently, the removal of riparian vegetation effects temperature most in small- and medium-sized streams, and least in large river systems. Sullivan and Adams (1990) suggest that riparian vegetation has a negligible effect on stream temperatures for streams that are 5th order or larger.

Although the effects of logging on stream temperatures within the logged area are well documented, the cumulative effects of temperature increases both downstream and over time are less well understood. Temperature data from Needle Branch in Oregon's Coast Range indicate that thermal regimes returned to near normal approximately seven years after logging and slash burning (Hall et al. 1987). In this case, alder replaced conifers as the dominant riparian vegetation and provided significant shade to this small stream. However, temperature increases in Carnation Creek showed no sign of diminishing eight years after logging and the author estimated that elevated stream temperatures were likely to persist for an additional decade or more (Holtby 1988). Similarly, in the higher elevation fir zone of the Cascade Range, the degree of shading may not reach prelogging levels for 40 years or more (Summers 1983). Thus the duration of temperature effects depends on the rate of recovery of riparian vegetation and the level of shading provided.

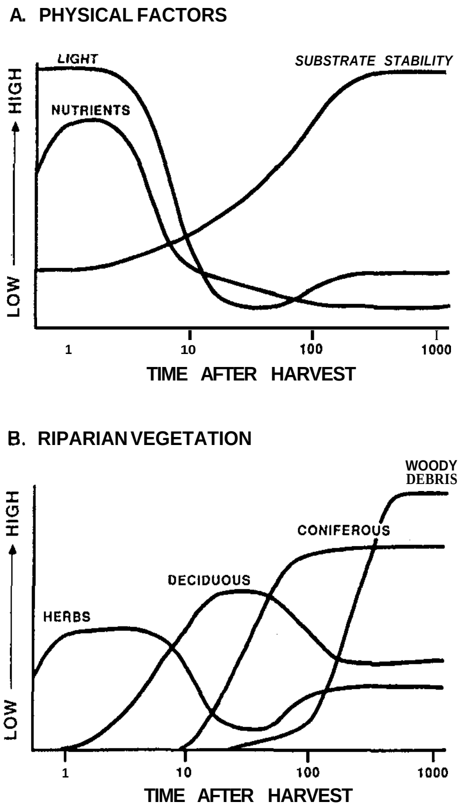
The cumulative effects of stream temperature changes downstream of logged areas are not well documented. As streams leave harvested areas and re-enter forested reaches, temperatures tend to decline as solar radiation is reduced. Similarly, small tributaries generally have a minor affect on the temperatures of larger streams which they enter (Caldwell et al. 1991). Sullivan and Adams (1990) have argued that in streams in western Washington, temperatures approach mean air temperatures at a "threshold distance" downstream from the watershed divide. The cumulative effects of warming of upstream tributaries may have little affect beyond this distance, though no analysis has been conducted to validate this hypothesis outside of western Washington (Beschta et al. 1995). In a study in the western Cascades of Oregon, Beschta and Taylor (1988) found that stream temperatures increased with increased logging and road building in the basin. As logging activity decreased in subsequent years, temperatures also declined, strongly suggesting that the cumulative effects of logging and road building were responsible for the previous temperature increases.

6.1.6 Effects on Nutrients

Forest practices can lead to changes in nutrient distribution and dynamics in upland areas, which in turn affect availability in streams. Nutrients are directly lost to the ecosystem through the removal of trees. Harvest intensity (i.e., proportion of forest canopy removed), type of harvest (logs or whole tree) and cutting frequency all affect the rate of nutrient removal from the system (Beschta et al. 1995). Despite the loss of nutrients stored in removed biomass, nutrients are generally more available to stream organisms in the years immediately following harvest (Figure 6-1). This results in part from the addition of slash to the forest floor (Frazer et al. 1990), accelerated decomposition of organic litter resulting from increased sunlight reaching the ground (Beschta et al. 1995), increased water availability for leaching of materials, and increased overland runoff and erosion that contributes unbound (nitrate and ammonium) and bound(orthophosphate) nutrients to the stream (Gregory et al. 1987). Where logging reduces riparian vegetation, nutrient supply to the stream (e.g., leaf litter and woody debris) may be reduced. As soils stabilize and revegetation occurs, the nutrient flux declines, though nutrients from herbaceous plants in the riparian zone add high quality materials that easily decompose. Over time herbs, shrubs, deciduous trees, and conifers provide allochthonous inputs for nutrient uptake (Figure 6-1).

Burning of slash, or the entire harvested area, can temporarily elevate the concentrations of nutrients entering the stream. Grier et al. (1989) suggests that fire effectively accelerates decomposition processes. If a fire is hot, however, much of the nitrogen is volatilized and lost to the system (Gessel and Cole 1973). In another study, potassium, phosphorus, calcium and magnesium increased by 2–8 times, while nitrogen decreased by two thirds following burning (Austin and Baisinger 1955). Herbicide treatments, like burning, can lead to short-term increases in nutrients as deciduous vegetation dies and decomposes.

The significance of forest harvest on nutrient losses depends on the mechanism causing the loss. The most significant losses result from tree removal. Leaching is not considered a major component of losses overall, accounting for less than 1% of losses from harvest. Losses because of volatilization resulting from fire can be much more significant (Beschta et al. 1995).


A. PHYSICAL FACTORS

B. RIPARIAN VEGETATION

Figure 6-1. Temporal patterns of physical factors (A) and riparian vegetation (B) after timber harvest (time is expressed as years on a logarithmic scale.) From Gregory et al. (1987). Reproduced with permission from the publisher.

6.1.7 Effects of Forest Chemicals

Fertilizers, herbicides, and insecticides are commonly used in forest environments to prepare sites for planting, to release and stimulate growth of conifers, and to control diseases and pests. In addition, fire retardants are used to halt the spread of wildfire on forest lands. All of these chemicals can affect salmonids through several direct and indirect pathways. Fertilizers, pesticides, and fire retardants that reach surface waters can be toxic to salmonids or may alter primary and secondary production, influencing the amount and type of food available to salmonids, Fertilizers and pesticides indirectly affect salmonids by influencing the rate of recovery of upland and riparian vegetation following logging, which in turn affects hydrologic processes, delivery of sediment and organic debris, heat transfer, nutrient cycling, and soil biota.

Contamination of surface waters by forest chemicals and the resultant risk of toxic effects on aquatic organisms depend on several factors, including the form and application rate of the chemical (and associated carriers), the application method (i.e., aerial versus ground spraying), soil type, weather conditions during and after application, and the retention of no-spray buffers in the riparian zone. The specific formulation determines the solubility of the chemical in water and its tendency to adsorb onto organic or inorganic matter within the soil, both of which determine the substance's mobility within the soil column. The persistence of forest chemicals in the environment varies. Some chemicals are highly volatile or are rapidly broken down through microbial activity or photodecomposition, whereas others persist for months or years. The volatility and adsorption of chemicals to soil particles are affected by temperature and soil moisture. If chemical application is followed by precipitation events, the likelihood that chemicals will reach surface waters increases. Contamination of surface waters also occurs when chemicals that have been applied to ephemeral channels are later mobilized during rainstorms. Aquatic organisms may be exposed to forest chemicals through direct contact with contaminated water, sediments, or food. No-spray buffers around streams (including ephemeral streams) and riparian areas substantially reduce the risk of contamination (Norris et al. 1991); however, toxic levels of chemicals may reach streams from storm runoff and wind drift even when best management practices are employed (e.g., Rashin and Graber 1993). Indirect effects of chemicals on watershed processes are largely unavoidable because change in vegetation is usually the desired outcome of such applications. Although both direct and indirect effects of forest chemicals on salmonids may be significant, it is important to note that less than 1% of total pesticides applied in the United States are used in forestry (Norris et al. 1991); thus, contamination from forest practices is likely minor compared to contamination from agricultural practices.

Fertilizers

Fertilizers are used in forest settings to replace nutrients lost during and after timber harvest and to accelerate growth of conifers. Application of fertilizers to a catchment typically results in increased concentrations of nutrients, particularly nitrogen, in streams. A recent review of effects of forest fertilization on water quality and aquatic biota indicates that urea application typically leads to elevated levels of urea-N, ammonia-N, and nitrate-N in surface waters (Bisson et al. 1992a). Urea-N usually dissipates within a few days, whereas ammonia-N may be elevated for months and nitrate-N for a year or more. The concentration of nitrogen within the stream depends on a number of factors, including the percentage of the watershed fertilized, the application rate, the drainage density (stream km relative to total watershed area), the width of unfertilized buffers along streams, and whether or not precipitation occurs following application. Although drinking water and aquatic standards are typically not exceeded with most applications, the elevation of nitrogen has the potential to promote growth of periphyton, which in turn may influence production of invertebrates and fishes. However, Bisson et al. (1992a) concluded that enhanced fish production because of forest fertilization has not been demonstrated in the Pacific Northwest. An indirect benefit of fertilizer applications is more rapid growth of vegetation within the catchment, which in turn accelerates the recovery of natural hydrologic regimes and sediment delivery rates.

Herbicides

In forest plantations, a wide variety of herbicides are used to control the invading hardwoods, herbaceous plants, and grasses to enhance the suitability of the area for re-establishment of desired tree species. Ten herbicides commonly used in forestry are 2,4-D, picloram, hexazinone, atrazine, imazapyr, triclopyr, forsamine, glyphosate, dalapon, and dinoseb. The behavior and toxicity of these substances is reviewed in detail in Norris et al. (1991) and Beschta et al. (1995), from whom much of the information below was excerpted.

The risk of toxicological effects of herbicides on salmonids is greatest when herbicides are directly applied to surface waters or reach surface waters by wind drift. Whether herbicides applied to upland forests will reach surface waters depends on their volatility, mobility in the soil, and persistence in the environment. Of the herbicides commonly used in forest applications, hexazinone, atrazine, imazapyr, and triclopyr are generally the most persistent, with soil half-lives of 2–6 months or more, depending on soil type. The half-lives of most other forest herbicides are generally from 2–5 weeks. Although there is substantial literature on the toxicity of various herbicides to salmonids, most of the available information comes from laboratory studies rather than the field. These laboratory studies focus on acute lethal doses (Reid 1993). Sublethal effects of herbicides on salmonids include reduced growth, decreased reproductive success, altered behavior, and reduced resistance to stress (reviewed in Beschta et al. 1995). Sublethal exposures of picloram were found to increase mortality by 70% in yearling coho exposed to seawater (Lorz et al. 1979). Information on effects of herbicides in aquatic invertebrates is also scarce. Hartman and Scrivener (1990) reported a **42%** reduction in the density of aquatic macroinvertebrates for 1.5 years following application of Roundup. These reductions were attributed to herbicide-induced irritation and drift of invertebrates coupled with high flows and decreases in substrate stability.

Herbicides used to release conifers from competing vegetation can accelerate the long-term recovery of upland and riparian areas. Over the short term, the elimination of deciduous vegetation can affect streams in several ways, both positive and negative. Herbicide applications in upland areas slow the recovery of vegetation, prolonging disruption to hydrologic and sediment delivery processes. Within the riparian zone, removal of deciduous vegetation increases solar radiation reaching streams, which stimulates algal production, potentially increasing the food base for invertebrates and fish. Delayed production of deciduous trees and accelerated growth of conifers reduces the delivery of leaves and intermediate-sized wood to streams over the short term, but increases the potential for recruitment of large coniferous wood over longer periods. Depending on whether temperature, spawning sites, cover, or food is limiting, these changes may initially hinder or aid salmonid production.

Insecticides

Insecticides are used both to prevent insect infestations and to control insect outbreaks once they have occurred. In general, insecticides are more toxic to fish and other aquatic biota than herbicides; however, they usually are applied at lower rates (Beschta et al. 1995). The greatest effect of insecticide on fish probably arises from effects on terrestrial and aquatic insects that form the salmonids' food base. Forest insecticides cause direct mortality to these insects or may stimulate catastrophic drift of aquatic invertebrates out of the affected stream reach. In addition, benthic algal communities in streams are frequently controlled by grazing invertebrates; consequently, the loss of invertebrates may release primary production, causing fundamental shifts in the trophic structure of streams. Norris et al. (1991) concluded that insecticides generally have shorter term effects on stream ecosystems than herbicides but that the effects may be more dramatic. Populations of invertebrates may take months to recover following insecticide applications, and full recovery of the invertebrate assemblage may take several years (Norris et al. 1991). Because salmonids in some forest streams may be food-limited, reductions in aquatic insect biomass and altered assemblage composition may result in reduced growth and numbers of salmonids. For example, Kingsbury (1983 in Norris et al. 1991) reported a decline in the growth rate of Atlantic salmon parr immediately following treatment with an insecticide; however, by the end of summer, fish in treated and untreated reaches were of similar size. Direct toxic effects may occur if salmonids consume drifting, pesticide-laden, aquatic organisms or terrestrial insects that fall into streams. Other indirect effects of insecticides on salmonid habitats are not well documented; however, protection of trees from insect pests may reduce the number of trees that die and fall into streams, thereby reducing recruitment of large woody debris.

Fire Retardants

The use of chemical fire retardants plays and important role in the suppression of wildfires in the west. Historically, a variety of chemicals have been used to suppress fires; however, ammonium-based retardants account for nearly all chemical retardants used today (Norris and Webb 1989). Although documentation of adverse effects of fire retardants on salmonids is scarce, quantities of retardant dropped during fires may be significant, and cases of fish mortality caused by retardants have been reported. For example, approximately 5.3 million liters of retardant were used to fight the Yellowstone fire of 1988, and at least two small fish kills (approximately 100 fish each) were reported (Schullery 1989). Fire retardant killed approximately 700 adult salmon, as well as a large number of juveniles, in an Alaskan stream (Hakala et al. 1971 in Norris and Webb 1989). Potential indirect effects of fire retardants on salmonids include mortality of invertebrates and eutrophication of downstream reaches (from phosphates). The extent of effects of retardants on aquatic ecosystems is influenced by application procedures (quantity applied, line of flight of aircraft relative to the stream), site characteristics (stream width-depth ratio, degree of canopy cover), and streamflow.

6.1.8 Effects on Physical Habitat Structure

Timber management activities have resulted in substantial modification of the physical characteristics of stream habitats throughout forested regions of the Pacific Northwest. Many of these changes have resulted from decreased recruitment of large woody debris (LWD) from the riparian zone and intentional removal of LWD from stream channels (Bisson et al. 1987; Maser et al. 1988; Hicks et al. 1991a). Removal of vegetation from the riparian zone has altered sources, mechanisms for delivery, distribution patterns, and stability of wood in stream channels (Bisson et al. 1987). Hardwoods have replaced conifers in many riparian areas following logging; woody debris produced by deciduous vegetation tends to be smaller, more mobile, and shorter-lived than that derived from conifers and, consequently, does not function as well in retaining sediment. The reduced supply of large woody debris decreases channel stability and eventually leads to loss of instream cover and pool habitat available for fish (Bisson et al. 1987). During the winter, salmonids have been reported to abandon reaches that are devoid of large wood (Tschaplinski and Hartman 1983). In addition, loss of large woody debris results in decreased retention of sediments, including gravels used by salmonids for spawning, as well as organic materials. The lack of debris also simplifies channel hydraulics, diminishing the heterogeneity that allows fishes to segregate among microhabitats. Loss of riparian vegetation also leaves banks unprotected, increasing bank erosion and reducing the formation of undercut banks that salmonids frequently use for cover. Hicks et al. (1991a) provide a good summary of short and long-term physical effects of forest practices on stream habitats (Table 6-4).

Excessive sedimentation resulting from logging and associated roads has also played a substantial role in altering salmonids habitats. Several studies have recorded increased levels of fine sediment in spawning gravels following logging (Cederholm and Reid 1987; Hall et al. 1987; Hartman et al. 1987); reductions in production of salmonids have been attributed in part to sedimentation of redds in each of theses instances, Fine sediments generated by logging and roading activities also can fill substrate interstices, decreasing the availability of cover for juvenile salmonids and altering primary production and invertebrate abundance (Cederholm and Reid 1987; Hicks et al. 1991a). Inputs of coarse sediments can fill pool habitats, resulting in channel shallowing and widening, and decreasing channel stability.

Culverts on logging roads create physical obstructions that fish must negotiate when migrating to and from the ocean and between summer or winter rearing habitats. Poorly designed and installed culverts act as barriers to both anadromous and resident salmonids. At culverts, excessive flow velocities, insufficient water, excessive culvert heights, and the absence of pools all can impede migration (Evans and Johnson 1980). Culverts also fail frequently when inappropriately designed and installed, resulting in mass wasting of road crossings. Road construction along streams, particularly where revetments are required, can constrain streamflow, thereby facilitating scouring of the channel bed. Dose and Roper (1994) found that channels widened as road density in catchments increased.

The cumulative effect of forest practices has been a reduction in the complexity of stream macro- and microhabitats. FEMAT (1993) documents substantial decreases in the number of large, deep pools in river systems west of the Cascade Range, which were attributed to loss of pool-forming structures (e.g., boulders, large wood), filling of pools with sediment, and loss of sinuosity of stream channels. Similar declines in pool frequency in river basins of eastern Oregon and Washington are reported in McIntosh et al. (1994a, 1994b). Reductions in pool habitat arise from the combined effects of loss of large woody debris, increased sediment inputs (which fill pools), and hydrologic changes that accompany land use. Although logging-related activities are not the only cause of these changes, they certainly play an important role. Effects of logging on physical habitat structure may persist for decades to a century or more. However, the causal linkages between land use, habitat development, and fish impacts are not always clear.

6.1.9 Effects on Stream Biota

A substantial volume of literature documents the effects of logging on aquatic biota (see reviews in Gregory et al. 1987; Hicks et al. 1991a; and Beschta et al. 1995). The response of aquatic communities to logging depends on a variety of factors, thus studies have sometimes produced seemingly contradictory results. Any of the following factors may influence the specific response of a given system: 1) species and stocks of fishes are diverse and adapted to local conditions; thus, the response may vary in different portions of each species' range; 2) physical and vegetative conditions, as well as logging methods, vary among regions such that impacts differ in magnitude, persistence, and ecological significance; 3) biotic interactions and long-range fish movements can mediate the effects of habitat alteration, such that most important biotic changes are indirectly and incompletely related to physical effects; 4) impacts of numerous independent factors can accumulate over time or space, or interact in either a compensatory or synergistic way, making ecological responses complex and difficult to predict; 5) dynamic, sometimes catastrophic natural events (e.g., large floods, changing oceanic conditions) create variable "baseline" conditions making it difficult to quantify additional variability caused by habitat alterations (Frissell 1991).

Within this context, some common patterns in the response of aquatic organisms to forest practices have been identified, based on both empirical evidence and theoretical expectations (e.g., the river continuum concept). In smaller streams, the removal of riparian

Table 6-4. Influences of timber harvest on physical characteristics of stream environments, potential changes in habitat quality, and resultant consequences for salmonid growth and survival. From Hicks et al. (1991a). Reproduced with permission from the publisher.

Forest Practice	Potential change in physical stream environment	Potential change in quality of salmonid habitat	Potential consequences for salmonid growth and survival
Timber harvest from streamside areas	Increased incident solar radiation	Increased stream temperature; higher light levels; increased autotrophic production	Reduced growth efficiency; increased susceptibility to disease; increased food production; changes in growth rate and age at smoking
	Decreased supply of large woody debris	Reduced cover; loss of pool habitat; reduced protection from peak flows; reduced storage of gravel and organic matter; loss of hydraulic complexity	Increased vulnerability to predation; lower winter survival; reduced carrying capacity; less spawning gravel; reduced food production; loss of species diversity
	Addition of logging slash (needles, bark, branches)	Short-term increase in dissolved oxygen demand; increased amount of fine particulate organic matter; increased cover	Reduced spawning success: short-term increase in food production; increased survival of juveniles
	Erosion of streambanks	Loss of cover along edge of channel; increased stream width; reduced depth	Increased vulnerability to predation; increased carrying capacity for age-0 fish, but reduced carrying capacity for age-1 and older fish
		Increased fine sediment in spawning gravels and food production areas	Reduced spawning success; reduced food supply
Timber harvest from hillslopes; forest roads	Altered streamflow regime	Short-term increase in streamflows during summer	Short-term increase in survival
		Increased severity of some peak flow events	Embryo mortality caused by bed-load movement
	Accelerated surface erosion and mass wasting	Increased fine sediment in stream gravels	Reduced spawning success: reduced food abundance; loss of winter hiding space
		Increased supply of coarse sediment	Increased or decreased rearing capacity
		Increased frequency of debris torrents; loss of instream cover in the torrent track; improved cover in some debris jams	Blockage to migrations; reduced survival in the torrent track; improved winter habitat in some torrent deposits
	Increased nutrient runoff	Elevated nutrient levels in streams	Increased food production
	Increased number of road crossings	Physical obstructions in stream channel; input of fine sediment from road surfaces	Restriction of upstream movement; reduced feeding efficiency
Scarification and slash burning (preparation of soil for reforestation)	Increased nutrient runoff	Short-term elevation of nutrient levels in streams	Temporary increase in food production
	Inputs of fine inorganic and organic matter	Increased fine sediment in spawning gravels and food production areas; short-term increase in dissolved oxygen demand	Reduced spawning success

vegetation increases light intensity, which stimulates the growth of benthic algae (Gregory 1980; Murphy et al. 1981; Shortreed and Stockner 1983; Murphy et al. 1986). In contrast, energy inputs from allochthonous sources decrease after harvest of riparian vegetation (Gregory et al. 1987; Bilby and Bisson 1992). Macroinvertebrate communities respond to these changes in food sources. Herbivorous invertebrates, particularly those that scrape algae from the substrate, are expected to become more abundant (but see Hawkins et al. 1982), while those species that feed on detritus (i.e., shredders, filterers, and collector-gatherers, sensu Merritt and Cummins 1984) typically decline in numbers (Hawkins et al. 1982; Beschta et al. 1995). The abundance of invertebrate predators has been shown to increase in response to increased secondary production in streams in the Oregon Cascades (Murphy et al. 1981; Hawkins et al. 1982). As riparian vegetation recovers, the amount of solar radiation reaching the channel diminishes, algal production decreases, and shredders and collector-gatherers begin to replace scrapers. It is important to note that these responses are likely typical only of streams where primary production is light-limited (i.e., small streams in dense forests); the response of invertebrates and juvenile salmonids to canopy removal in more open, eastside systems may differ.

The general pattern of change in aquatic communities in response to changing energy sources caused by logging of riparian vegetation can be confounded by other simultaneous changes in habitat conditions. Silt deposited from mass failures and surface erosion can affect invertebrate production as gravel interstices are filled by silt, and algae are buried or abraded (Beschta et al. 1995). In these instances, invertebrate assemblages are typically characterized by high numbers of a few tolerant, colonizing species (Newbold et al. 1980; Murphy et al. 1981; Hawkins et al. 1982; Lamberti et al. 1991). Loss of substrate complexity, including large woody debris, also tends to decrease the diversity of aquatic invertebrates. Similarly, application of insecticides and herbicides may have substantial and long-lasting effects on invertebrate community structure, with stoneflies, mayflies, and caddisflies all being particularly sensitive (reviewed in Beschta et al. 1995). Gregory et al. (1987) suggest that an overall pattern of increased production of a few taxa accompanied by a reduction in biodiversity may be common to all invertebrate trophic levels in streams that have been simplified through forest practices.

All of the logging-induced changes in physical and biological characteristics discussed in preceding sections interact to influence the composition and diversity of fish populations and communities; however, few studies in the Pacific Northwest have been designed to address specific relationships between changes in habitat attributes and structure of fish assemblage (Bisson et al. 1992). In addition, changes in one habitat attribute that benefit salmonid productivity (e.g., increased light and primary production) may be compensated by other adverse effects (e.g., loss of rearing habitat, sedimentation, excessive temperatures, change in timing of life-history events), which may not be manifest until subsequent life stages. Similarly, most studies of effects of logging in the Pacific Northwest have been conducted in relatively wet, forested ecosystems of the Coast Range and Western Cascades. In eastside systems, canopy removal, loss of physical structure, and increases in sediment loading may have greater ecological impacts on salmonids because different factors may limit production (e.g., summer temperatures, pool habitats).

Studies in the Cascades and Coast Ranges of Oregon and northern California indicated higher densities and biomass of salmonids in recently clear-cut reaches compared to shaded old-growth and second-growth reaches (Murphy et al. 1981; Hawkins et al. 1983); however, no analysis of age structure was provided. In western Washington, Bisson and Sedell (1984) found that total salmonid biomass was greater in streams that had been logged and cleaned (i.e., large woody debris removed), but that populations were dominated by underyearling trout, with proportionately fewer age I and older trout. These differences were attributed in part to a higher frequency of riffles (favored by underyearlings) and a lower frequency of pools (favored by older trout) in clear-cut reaches. Subsequent studies suggest that juvenile coho salmon and older trout, which prefer pool habitats, are adversely affected by conversion of pools to riffles (Bisson et al. 1992). In a long-term study of the effects of logging in an Oregon watershed (Alsea Watershed Study), numbers of fry migrating from a clear-cut section decreased by more than 50%, while prelogging and postlogging numbers from a patch-cut watershed and an unharvested watershed were not significantly different (Hall et al. 1987). Declines in the clear-cut watershed were attributed to a reduction in gravel quality from increased fines that led to a decrease in survival from egg deposition to emergence. In the same study, Moring and Lantz (1975) found reductions in late-summer densities of cutthroat trout in the clear-cut watershed following logging, but no reductions in the other two watersheds. Hartman and Scrivener (1990) reported that numbers of steelhead smolts declined, but cutthroat numbers remained constant following logging of 41% of the Carnation Creek watershed in British Columbia. In southeast Alaska, summer abundance of coho salmon increased following canopy removal; however, in winter parr densities were lower in clear-cut reaches than old growth reaches, apparently because of a lack of large woody debris and undercut banks (Heifitz et al. 1986).

Other general effects of logging on fishes include decreased growth efficiency, reduced survival to emergence, increased susceptibility to disease, increased vulnerability to predation, lower winter survival, blockages to migration from poorly designed culverts, changes in the age-structure of fish populations, reduced development time of embryos, and altered timing of life-history events (Hicks et al. 1991a). Scrivener and Brownlee (1989) reported reductions in survival to emergence of coho and chum salmon of approximately 45% as a result of increased fine sediment in spawning gravels following logging of the Carnation Creek watershed. In the same watershed, Holtby et al. (1989) reported that increases in water temperatures following logging resulted in earlier outmigration of coho and chum salmon fry and earlier emigration by coho salmon smolts to the ocean. The change in migration timing of coho salmon smolts is believed to have reduced ocean survival. Furthermore, outmigrating coho smolts were evenly divided between 1- and 2-year old fish in years preceding logging, but dominated by l-year old fish after logging. Such changes increase the vulnerability of specific year classes to environmental fluctuations in both the freshwater and marine environments. The Alsea Watershed study documented an increase in the number of early (November–January) coho migrants (Hicks et al. 1991a); whether the change was due to temperature-induced acceleration of growth or loss of rearing habitats remains uncertain. Nevertheless, these results suggest that small increases in temperature (1–2°C) can result in significant shifts in the timing of important life history events.

Finally, roads constructed for timber harvest may indirectly affect salmonids by increasing public access to previously remote locations. Angling pressure generally decreases with increasing distance from access roads; consequently, increased mortality from angling may accompany habitat degradation.

6.2 Grazing

Livestock grazing represents the second most dominant land use in the Pacific Northwest, following timber production. In Oregon, Washington, and Idaho combined, over 22.9 million hectares (56.5 million acres) of grassland and desert shrubland, approximately 3.2 million hectares (8 million acres) of nonfederal forest land, and an undetermined amount of Federal forest land are grazed by cattle and sheep (Pease 1985). This acreage represents approximately 41% of the total land base. Rangelands are fairly evenly divided between Federal and nonfederal lands; Federal rangelands total approximately 12.3 million hectares (30.5 million acres) (excluding Federal forest lands that are grazed), and nonfederal rangelands total 13.8 million hectares **(34** million acres). Estimates from 1987 indicate that 4.76 million cattle and 0.87 million sheep were produced for sale in Idaho, Oregon, and Washington (Pease 1993). The majority of rangelands in Washington and Oregon lie east of the Cascade Range (Palmisano et al. 1993a), but livestock are also concentrated in the Willamette Valley and Puget Lowlands west of the Cascades, as well as in coastal valleys of Washington and Oregon. No estimates of rangeland area in northern California were available. However, from 1966–1980 California produced an average of approximately 4.75 million cattle and 1.0 million sheep annually (Hornbeck et al. 1983), a number comparable to the livestock production of Oregon, Washington, and Idaho combined. The largest concentrations of livestock in California within the current range of the Pacific salmon occur in the Sacramento-San Joaquin Valley, the Sierra Nevada and Coast Range foothill regions, and coastal valleys of northern California.

Livestock grazing in the West was already heavy by the mid-to-late 1800s. In 1898, the National Academy of Sciences prepared a report for the Interior Department alleging significant destruction by unregulated grazing in national Forest Reserves (Irwin et al. 1994). By the late 1920s, concern about deterioration of rangelands on national forests was growing (Platts 1991; Heady and Child 1994). In the 1930s the Forest Service documented widespread degradation of rangeland conditions, concluding that overgrazing had destroyed more than half of all rangelands and that 75% of remaining rangelands were degraded (Heady and Child 1994). Concern for rangeland conditions prompted Congress to enact the Taylor Grazing Act of 1934, which established 80 million acres of land in grazing districts to be administered by the **U.S.** Grazing Service, later to become the U. **S.** Bureau of Land Management (BLM) (Platts 1991; Wilkinson 1992). The percentage of total rangeland in "poor" condition decreased from 36% in 1936 to 18% in 1984, suggesting some improvement in overall range condition (Heady and Child 1994). However, recent reports have indicated that most riparian areas remain in fair-to-poor condition (Chaney et al. 1990; GAO 1991). Thus, while upland conditions appear to be improving, riparian areas continue to be degraded. In 1991, BLM began a program to improve riparian management, with a goal of restoring 75% or more of riparian areas to properly functioning condition by 1997 (Barrett et al. 1993).

Despite the generally poor condition of most riparian areas, the potential for restoring those areas damaged by grazing is arguably greater than for those affected by other activities (Behnke 1977; Platts 1991). Recovery of grasses, as well as willows and other woody species, can occur within a few years when grazing pressure is reduced or eliminated (Elmore and Beschta 1987; Platts 1991; Elmore 1992). Restoration of fully functioning riparian areas that support a variety of plant species, including older forests of cottonwood and other large tree species, will take considerable time. Nevertheless, many important riparian functions—shading, bank stabilization, sediment and nutrient filtering, and allochthonous inputs—may be rapidly restored to the benefit of salmonids, provided the stress of grazing is alleviated and prior damage has not been too severe.

6.2.1 Effects on Vegetation

Heavy grazing around the turn of the century had significant and widespread effects, many of which persist today, on upland and riparian vegetation. Rangelands have experienced decreases in the percentage of ground covered by vegetation and associated organic litter (Heady and Child 1994). Species composition of plants in upland areas have shifted from perennial grasses toward non-native annual grasses and weedy species (Heady and Child 1994). East of the Cascade Range, upland sites that once supported plant associations of Idaho fescue are now devoid of native bunchgrasses, which have been replaced with tarweed, gumweed, and other noxious plants (Johnson et al. 1994). In riparian areas, willow, aspen, sedge, rush, and grass communities have been reduced or eliminated and replaced with annual grasses or sagebrush. Diaries of early trappers in eastern Oregon noted that grasses were as high as seven feet (Wilkinson 1992) and that streams were well lined with willows, aspen, and other woody vegetation (Elmore 1992). In eastside meadows, alteration of the vegetation has been so pervasive that little is known about the native vegetation that once inhabited riparian meadow communities. Currently, these meadows are dominated by Kentucky bluegrass, big sagebrush, and annual brome grasslands (Johnson et al. 1994). Fleischner (1994) recently reviewed the literature and found numerous examples of changes in species composition, diversity, and richness associated with livestock grazing or removal of livestock in western States (Table *6-5).*

Table 6-5. Deleterious effects of livestock grazing on plant communities in western North America. From Fleischner (1994). Reprinted with permission of Blackwell Scientific Publications, Inc.

Habitat	Location	Effect	Authority
Sonoran desert scrub	Arizona	Perennial grasses and Krameria (palatable shrub) showed dramatic density decreases with grazing	Blydenstein et al. (1957)
Mojave desert scrub	California	60% reduction in above-ground biomass of annuals, 16%–29% decrease' in cover of perennial shrubs with grazing	Webb and Stielstra (1979)
Sagebrush desert	Idaho	Grazed site had one-third species richness of ungrazed site	Reynolds and Trost (1980)
Desert grassland	New Mexico	Grass density increased by 110% after 30 years of protection from grazing	Gardner (1950)
Semidesert grassland	Arizona	Species richness increased as did canopy cover for midgrass, shortgrass, shrub, and forb groups after removal of livestock	Brady et al. (1989)
Semidesert grassland	Arizona	Woody plants significantly more abundant after removal of livestock	Bock et al. (1984)
Ponderosa pine forest	Washington	Decreased species richness on grazed sites	Rummell (1951)
Mountain canyon	Utah	Absence or near absence of 10 grass species on grazed sites	Cottam and Evans (1945)
Riparian	Oregon	Species richness increased from 17 to 45 species 9 years after removal of livestock	Winegar (1977)
Riparian	Arizona	Herbaceous cover of grazed plot less than half that of ungrazed plot	Szaro and Pase (1983)
Riparian	Colorado	Shrub canopy coverage increased 5.5 times, willow canopy coverage 8 times after removal of livestock	Schulz and Leininger (1990)

Much early alteration of rangelands was by settlers who engaged in widespread clearing of grasslands and riparian forests to grow crops, build houses, obtain fuelwood, and increase availability of land for domestic animals (Heady and Child 1994). Conversion of lands for the purpose of livestock production continues today. Woody shrubs and trees are sometimes removed by chaining and cabling—uprooting of vegetation with anchor chains or cables stretched between tractors—for the purpose of increasing grass production (Heady and Child 1994). Removal of woody shrubs through chemical application or by mechanical means is also a common practice in range management. In addition, suppression of fire on rangelands is responsible for changes in upland vegetation, including encroachment by juniper in many areas of eastern Oregon and Washington (Miller et al. 1989a).

Cattle and sheep affect vegetation primarily through browsing and trampling. Grazing animals are selective in what they eat; consequently, preferred vegetation types are generally removed first, followed by less palatable species. Heavy, continual grazing causes plants to be partially or wholly defoliated, which can reduce biomass, plant vigor, and seed production (Kauffman 1988; Heady and Child 1994). Selection of specific plant species may allow other taxa to dominate (Kauffman and Krueger 1984; Fleischner 1994). Vegetation may also be lost or damaged through trampling, which tears or bruises leaves and stems, and may break stems of woody plants. Regeneration of some woody vegetation, such as willow, cottonwood, and aspen, is inhibited by browsing on seedlings (Fleischner 1994). Vegetation may also be directly lost when buried by cattle dung. In a dairy pasture, MacDiarmid and Watkin (1971) found that 75% of grasses and legumes under manure piles were killed.

Livestock grazing also influences vegetation through modification of soil characteristics. Hooves compact soils that are damp or porous, which inhibits the germination of seeds and reduces root growth (Heady and Child 1994). Changes in infiltration capacity associated with trampling may lead to more rapid surface runoff, lowering moisture content of soil and hence the ability of plants to germinate or persist (Heady and Child 1994); however, in some instances, trampling may break up impervious surface soils, allowing for greater infiltration of water and helping to cover seeds (Savory 1988 in Heady and Child 1994). Soils in arid and semi-arid lands have a unique microbiotic surface layer or crust of symbiotic mosses, algae, and lichens that covers soils between and among plants. This "cryptogamic crust" plays important roles in hydrology and nutrient cycling (see Sections 6.2.3 and 6.2.6) and is believed to provide favorable conditions for the germination of vascular plants (Fleischner 1994). Trampling by livestock breaks up these fragile crusts, and reformation may take decades. Anderson et al. (1982) found recovery of cryptogamic crusts took up to 18 years in ungrazed exclosures in Utah. Finally, livestock indirectly affect plant species composition by aiding the dispersion and establishment of non-native species; seeds may be carried on the fur or in the dung of livestock (Fleischner 1994).

The effects of livestock grazing on vegetation are especially intense in the riparian zone because of the tendency for livestock to congregate in these areas. Gillen et al. (1984) found that 24%–47% of cattle in two pastures in north-central Oregon were observed in riparian meadows constituting only 3%–5% of the total land area. Roath and Krueger (1982) reported that riparian meadows that constituted only 1%–2% of the total land area accounted for 81% of the total herbaceous biomass removed by livestock. Similar preferences for riparian areas have been observed elsewhere in the west (reviewed in Kauffman and Krueger 1984; Fleischner 1994). Cattle and sheep typically select riparian areas because they offer water, shade, cooler temperatures, and an abundance of high quality food that typically remains green longer than in upland areas (Kauffman and Krueger 1984; Fleischner 1994; Heady and Child 1994). In mountainous terrain, the preference of cattle and sheep for the riparian zone also appears related to hillslope gradient (Gillen et al. 1984). Heady and Child (1994) suggest that cattle avoid slopes greater than 10%–20%. The intensity of use by livestock in riparian zones exacerbates all of the problems noted above and generates additional concerns. Alteration of flow regime, changes in the routing of water, and incision of stream channels can lead to reduced soil moisture in the floodplain. Many types of riparian vegetation are either obligate or facultative wetland species that are adapted to the anaerobic conditions of permanently or seasonally saturated soils. Stream downcutting and the concomitant lowering of the water table can lead to encroachment of water-intolerant species such as sagebrush and bunchgrasses into areas formerly dominated by willows, sedges, rushes, and grasses (Elmore 1992). In addition, flood events may be an important mechanism for seed dispersal throughout the floodplain for woody plants, a function that is diminished as channels are incised.

6.2.2 Effects on Soils

Rangeland soils are frequently compacted by livestock. The degree of soil compaction depends on soil characteristics, including texture, structure, porosity, and moisture content (Platts 1991; Heady and Child 1994). As a general rule, soils that are high in organic matter, porous, and composed of a wide range of particle sizes are more easily compacted than other soils. Similarly, moist soils tend to be more susceptible to compaction than dry soils, although extremely wet soils may give way and then recover following trampling by livestock (Clayton and Kennedy 1985). The result of soil compaction is an increase in bulk density (specific gravity) in the top 5–15 cm of soil as pore space is reduced. Because of the loss of pore space, infiltration is reduced and surface runoff is increased, thereby increasing the potential for erosion (see Section 6.2.4). The available studies indicate that compaction generally increases with grazing intensity, but that site-specific soil and vegetative conditions are important in determining the response of soils to this grazing activity (reviewed in Kauffman and Krueger 1984; Heady and Child 1994).

Trampling by livestock may also displace or break up surface soils. In instances where surface soils have become impervious to water, light trampling may increase the soil's ability to absorb water. On the other hand, loosening soils makes them more susceptible to erosion. Heavily pulverized soil (dust) may become hydrophobic, reducing infiltration and increasing surface runoff. In arid and semi-arid climates, the cryptogamic crust has been shown to increase soil stability and water infiltration (Loope and Gifford 1972; Kleiner and Harper 1977; Rychert et al. 1978). Disruption of the cryptogamic crust may thus have long-lasting effects on erosional processes.

Livestock also alter surface soils indirectly by removing ground cover and mulch, which in turn affects the response of soils to rainfall. Kinetic energy from falling raindrops erodes soil particles (splash erosion), which may then settle in the soil interstices resulting in a relatively impervious surface. Livestock grazing can increase the percentage of exposed soil and break down organic litter, reducing its effectiveness in dissipating the energy of falling rain.

6.2.3 Effects on Hydrology

Grazing modifies two fundamental hydrologic processes, evapotranspiration and infiltration, that ultimately affect the total water yield from a watershed and the timing of runoff to streams. Loss of upland and riparian vegetation results in reduced interception and transpiration losses, thereby increasing the percentage of water available for surface runoff (Heady and Child 1994). Shifts in species composition from perennials to annuals may also reduce seasonal transpiration losses. Reductions in plant biomass and organic litter can increase the percentage of bare ground and can enhance splash erosion, which facilitates clogging of soil pores and decreases infiltration. Similarly, soil compaction reduces infiltration. Rauzi and Hanson (1966) report higher infiltration rates on lightly grazed plots, compared to moderately and heavily grazed plots in South Dakota. Similar experiments in northeastern Colorado showed reductions in infiltration in heavily grazed plots, but no differences between moderately and lightly grazed plots (Rauzi and Smith 1973). Johnson (1992) reviewed studies related to grazing and hydrologic processes and concluded that heavy grazing nearly always decreases infiltration, reduces vegetative biomass, and increases bare soil.

Decreased evapotranspiration and infiltration increases and hastens surface runoff, resulting in a more rapid hydrologic response of streams to rainfall. Some authors have suggested that the frequency of damaging floods has increased in response to grazing; however, there remains uncertainty about the role of grazing in mediating extreme flow events (reviewed in Fleischner 1994).

Reduced stability of streambanks associated with loss of riparian vegetation can lead to channel incision or "downcutting" during periods of high runoff. In naturally functioning systems, riparian vegetation stabilizes streambanks, slows the flow of water during high flow events, and allows waters to spread out over the floodplain and recharge subsurface aquifers (Elmore 1992). Moreover, riparian vegetation facilitates sediment deposition and bank building, increasing the capacity of the floodplain to store water, which is then slowly released as baseflow during the drier seasons (Elmore and Beschta 1987). Downcutting effectively separates the stream channel from the floodplain, allowing flood waters to be quickly routed out of the system and leading to lowering of the water table (Platts 1991; Elmore 1992; Armour et al. 1994). Consequently, summer streamflows may decrease even though total water yield increases in response to vegetation removal (Elmore and Beschta 1987). Li et al. (1994) found that streamflow in a heavily grazed eastern Oregon stream became intermittent during the summer, while a nearby, well vegetated reference stream in a similar-sized watershed had permanent flows. They suggested that the difference in flow regimes was a consequence of diminished interaction between the stream and floodplain with resultant lowering of the water table.

6.2.4 Effects on Sediment Transport

Livestock presence in the riparian zone increases sediment transport rates by increasing both surface erosion and mass wasting (Platts 1991: Marcus et al. 1990; Heady and Child 1994). Devegetation and exposure of soil by grazing facilitates detachment of soil particles during rainstorms, thereby increasing overland sediment transport. Rills and gullies often form in areas denuded by livestock trails or grazing, resulting in increased channelized erosion (Kauffman et al. 1983). As gullies expand and deepen, streams downcut, the water table drops, and sediments are transported to depositional areas downstream (Elmore 1992; Fleischner 1994; Henjum et al. 1994). Stream downcutting leads to further desertification of the riparian area and promotes soil denudation and the establishment of xeric flora. This in turn increases the potential for soil erosion. Some evidence suggests that significant channel downcutting in the Southwest occurred prior to the introduction of livestock (Karlstrom and Karlstrom 1987 in Fleischner 1994); however, studies in eastern Oregon and northern California implicate livestock as a major cause of downcutting (Dietrich et al. 1993; Peacock 1994).

Mass wasting of sediment occurs along stream banks where livestock trample overhanging cut banks (Behnke and Zarn 1976; Platts and Raleigh 1984; Fleischner 1994). Grazing also removes vegetation that stabilizes streambanks (Platts 1991). Where banks are denuded, undercutting and sloughing occurs, increasing sediment loads, filling stream channels, changing pool-riffle ratios, and increasing channel width (Platts 1981 in Fleischner 1994).

6.2.5 Effects on Thermal Energy Transfer and Stream Temperature

Riparian vegetation shades streams and thereby regulates stream temperatures. On rangelands east of the Cascades, black cottonwood, mountain alder, and quaking aspen are the dominant deciduous tree species in natural communities, whereas west of the Cascades, black cottonwood, red alder, and big leaf maple are dominant (Kauffman 1988). Shrubby vegetation, such as willows, may also be important sources of shade along smaller streams and in mountainous areas (Henjum et al. 1994), and even tall grasses can provide some measure of shade along narrow first- and second-order streams (Platts 1991).

The removal of riparian vegetation along rangeland streams can result in increased solar radiation and thus increased summer temperatures. Li et al. (1994) noted that solar radiation reaching the channel of an unshaded stream in eastern Oregon was six times greater than that reaching an adjacent, well shaded stream and that summer temperatures were 4.5°C warmer in the unshaded tributary. Below the confluence of these two streams, reaches that were unshaded were significantly warmer than shaded reaches both upstream and downstream. A separate comparison of water temperatures at two sites of similar elevation in watersheds of comparable size found temperature differences of 11°C between shaded and unshaded streams (Li et al. 1994). Warming of streams from loss of riparian vegetation is likely widespread east of the Cascades and may be particularly acute because of low summer flows and a high percentage of cloud-free days.

The effects of riparian canopy in winter on stream temperatures are less well understood and various studies have shown increases, decreases, and no change in water temperature following removal of riparian canopy (reviewed in Beschta et al. 1987). Riparian cover can inhibit energy losses from evaporation, convection, and long-wave radiation during the winter, and several authors have suggested that removal of vegetation can increase radiative heat loss and facilitate the formation of anchor ice (Beschta et al. 1991; Platts 1991; Armour et al. 1994). This is most likely to occur in regions where skies are clear on winter nights and where snow-cover is inadequate to blanket and insulate streams (Beschta et al. 1987), primarily in mountainous regions and east of the Cascades and Sierra Nevada.

Alteration of stream temperature processes may also result from changes in channel morphology. Streams in areas that are improperly grazed tend to be wider and shallower than in ungrazed systems, exposing a larger surface area to incoming solar radiation (Bottom et al. 1985; Platts 1991). Wide, shallow streams heat more rapidly than narrow, deep streams (Brown 1980). Similarly, wide, shallow streams may cool more rapidly, increasing the likelihood of anchor ice formation. Reducing stream depth may expose the stream bottom to direct solar radiation, which may allow greater heating of the substrate and subsequent conductive transfer to the water.

6.2.6 Effects on Nufrienfs and Other Solutes

Livestock activities can directly affect nutrient dynamics through several mechanisms, The removal of riparian vegetation by grazing reduces the supply of nutrients provided by organic leaf litter. Livestock also redistribute materials across the landscape. Because riparian areas are favored by cattle and sheep, nutrients that have been ingested elsewhere on the range tend to be deposited in riparian zones or near other attractors, such as salt blocks (Heady and Child 1994). The deposition of nutrients in riparian areas increases the likelihood that elements such as nitrogen and phosphorous will enter the stream. Nutrients derived from livestock wastes may be more bioavailable than those bound in organic litter. Elimination of the cryptogamic crust by livestock may also alter nutrient cycling in arid and semi-arid systems. These microbiotic crusts perform the majority of nitrogen fixation in desert soils (Rychert et al. 1978). Loss of these crusts can reduce the availability of nitrogen for plant growth, potentially affecting plant biomass in uplands (Fleischner 1994).

Riparian areas play a major role in regulating the transportation and transformation of nutrients and other chemicals (see Section 3.9.6). As stream channels incise and streams are separated from their floodplains, soil moisture is reduced, which in turn alters the quantity and form of nutrients and their availability to aquatic communities. In the anaerobic environments of saturated soils, microbial activity transforms nitrate nitrogen (NO,) into gaseous nitrous oxide (N_2O) and elemental nitrogen (N_2) that are liberated to the atmosphere (Figure 6-2) (Green and Kauffman 1989). Under drier soil conditions (oxidizing environments), denitrification does not occur and nitrate-nitrogen concentrations in the soil increase. Because nitrate is negatively charged, it is readily transported by subsurface flow to the stream channel (Green and Kauffman 1989). Thus by altering the hydrologic conditions in the riparian zone, grazing can increase the amount of nitrate nitrogen released to streams. Excessive nitrate concentrations facilitate algal growth, increase turbidity, and in some cases cause oxygen depletion because of increased biochemical oxygen demand.

The form of other elements including manganese, iron, sulfur, and carbon also depends on the redox potential of soils. In their reduced form, manganese, iron, and sulfur can be toxic to plants at high concentrations (Green and Kauffman 1989). Obligate and facultative wetland plant species have special adaptations for coping with these reduced elements that allow them to survive where more xeric plants cannot. Thus, changes in hydrologic condition caused by downcutting can modify the form of elements available to plants, thereby altering competitive interactions among plants and changing riparian plant communities.

6.2.7 Effects of Vegetation Management

Fertilizers, herbicides, mechanical treatments, and prescribed fire are commonly used in rangeland management to alter vegetation in favor of desired species. In principle, the potential effects of these activities on salmonids and their habitats are no different than similar activities in forested environments; however, because the physical and biological processes that regulate the delivery of water, sediments, and chemicals to streams differ on forests and rangelands, so may the response of aquatic ecosystems.

Fertilizers are used on rangelands to increase forage production, improve nutritive quality of forage, and enhance seedling establishment, although the high costs and varied results have led to a decline in fertilizing rangeland in the past 20 years (Heady and Child 1994). Fertilizers that reach streams through direct application or runoff can adversely

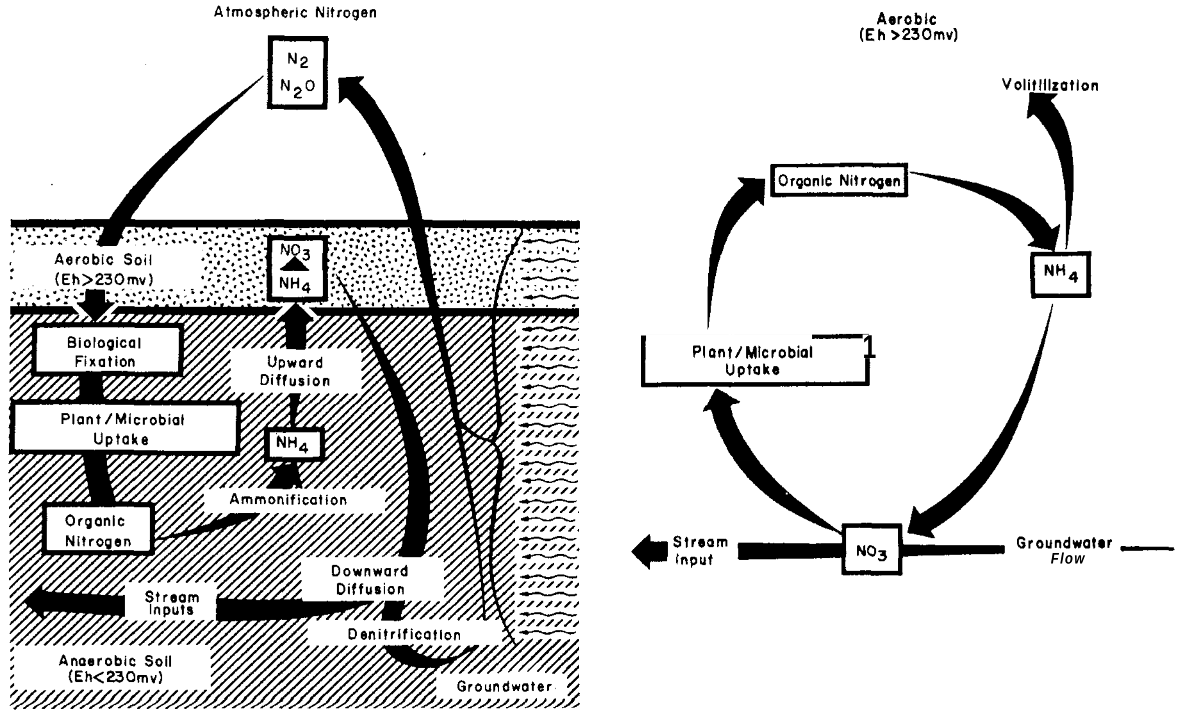

Atmospheric Nitrogen
N2
N2O
Aerobic Soil
(Eh>230mv)
NO3
NH4
Biological Fixation
Upward Diffusion
Plant/Microbial Uptake
NH4
Ammonification
Organic Nitrogen
Downward Diffusion
Stream Inputs
Denitrification
Anaerobic Soil
(Eh<230mv)
Groundwater
Aerobic
(Eh >230mv)
Volitilization
Organic Nitrogen
NH4
Plant/Microbial Uptake
Stream Input
NO3
Groundwater Flow

Figure 6-2. Nitrogen cycling pathways in undisturbed (left) and disturbed (right) riparian zones of northeastern Oregon, as indicated by redox potential (Eh). From Green and Kauffman (1989). Reproduced with permission from the principal author.

affect water quality. Nutrient enrichment (especially nitrogen) promotes algal growth, which in turn can lead to oxygen depletion as algae die and decompose. Conversely, fertilizer applied to rangelands may reduce sedimentation, hydrologic, and temperature effects by stimulating recovery of vegetation, including woody riparian shrubs. Herbicides are typically used to target unpalatable or noxious weeds that compete with desired forage species. Many of the herbicides commonly used in forestry (e.g., 2,4-D, picloram, glyphosate, tricopyr) are used in range management as well, although other highly selective herbicides may be used to control particular weeds common to rangelands, including unpalatable woody shrubs. Direct toxic effects on aquatic biota may occur where herbicides are applied directly to stream channels; however, risks of contamination can be minimized if adequate no-spray buffers are maintained (Heady and Child 1994). Herbicide applications to upland areas may decrease total groundcover, increasing the potential for surface erosion. In the riparian zone, use of herbicides may reduce production of deciduous trees and shrubs, opening streams to greater direct solar radiation, which in turn leads to elevated stream temperatures and increased algal production. These conditions can lead to insufficient nighttime dissolved oxygen concentrations and afternoon gas supersaturation. The loss of riparian vegetation also decreases the amount of organic litter and large wood delivered to streams. Furthermore, without the root structure of woody vegetation, banks are prone to collapse, increasing sedimentation and reducing the amount of cover for fish.

The influence of mechanical treatment and prescribed fire on aquatic ecosystems in rangelands depends on the type and intensity of disturbance. The use of tractors with dozer blades, brush rakes, cables, or rolling cutters for vegetation removal all can lead to compaction of rangeland soils (Heady and Child 1994) and thereby increase surface runoff and erosion. Disking of soils may break up relatively impervious soils, allowing greater infiltration of water; however, unless the area is rapidly revegetated, raindrop splash on exposed soils is likely to facilitate surface erosion and increase sediment delivery to streams. Disking and dozer use also rearrange soil layers, mixing topsoil with woody debris, which may affect re-establishment of vegetation. Positive effects of mechanical vegetation removal are also possible, Removal of vegetation with high evapotranspiration rates (e.g., juniper woodlands that have encroached because of grazing and lack of wildfire) may potentially increase the amount of water available during the summer, although documentation of this effect is poor. Prescribed fire is most likely to affect aquatic ecosystems through increased surface runoff and erosion resulting from the removal of vegetation and formation of hydrophobic soils.

In summary, manipulations of vegetation on rangelands can influence salmonid habitats through both direct and indirect pathways. These changes may harm or benefit salmonids depending on whether temperature, spawning sites, cover, or food limit the production of salmonids. Salmonid abundance will decrease if the increased invertebrate production is offset by undesirable alterations in the benthos assemblage to less nutritious species, reduced cover, increased sedimentation, and lower water quality.

6.2.8 Effects on Physical Habitat Structure

Livestock-induced changes in physical structure within streams result from the combined effects of modified hydrologic and sediment transport processes in uplands and the removal of vegetation within the riparian zone. Platts (1991) and Elmore (1992) review effects of grazing on channel morphology and are the sources of most information presented below. Loss of riparian vegetation from livestock grazing generally leads to stream channels that are wider and shallower than those in ungrazed or properly grazed streams (Hubert et al. 1985; Platts and Nelson 1985a, 1985b in Marcus et al. 1990). Loss of riparian root structure promotes greater instability of stream banks, which reduces the formation of undercut banks that provide important cover for salmonids (Henjum et al. 1994). Furthermore, the increased deposition of fine sediments from bank sloughing may clog substrate interstices, thereby reducing both invertebrate production and the quality of spawning gravels.. Over the long-term, reductions in instream wood diminish the retention of spawning gravels and decrease the frequency of pool habitats. In addition, the lack of structural complexity allows greater scouring of streambeds during high-flow events, which can reduce gravels available for spawning and facilitate channel downcutting. Figure 6-3 illustrates the characteristics of vegetation in functional and dysfunctional riparian zones on rangelands, and the channel modifications that typically result.

6.2.9 Effects on Stream Biota

As with forest practices, removal of riparian vegetation by livestock can fundamentally alter the primary source of energy in streams. Reductions in riparian canopy increase solar radiation and temperature, which in turn stimulates the production of periphyton (Lyford and Gregory 1975). In a study of seven stream reaches in eastern Oregon, Tait et al. (1994) reported that thick growths of filamentous algae encrusted with epiphytic diatoms were found in

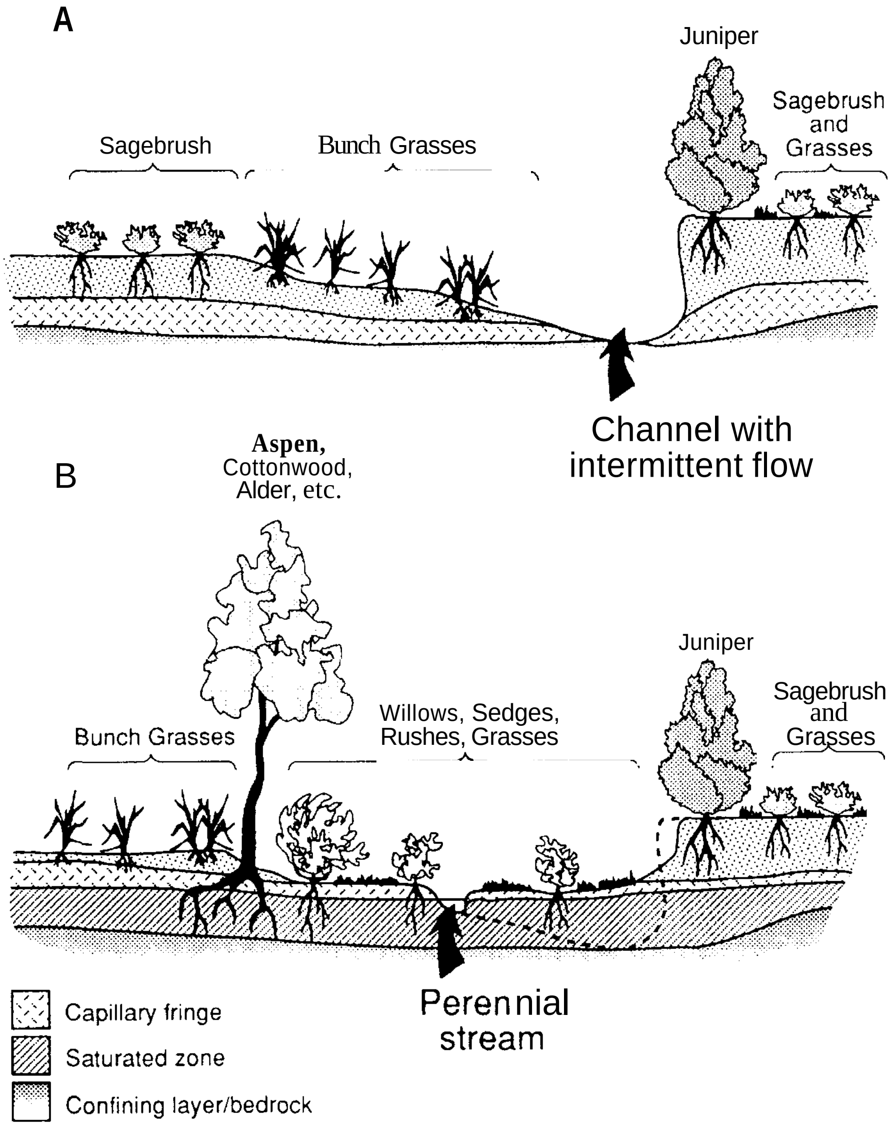

A
Juniper
Sagebrush and Grasses
Sagebrush
Bunch Grasses
Channel with intermittent flow
B
Aspen, Cottonwood, Alder, etc.
Juniper
Sagebrush and Grasses
Willows, Sedges, Rushes, Grasses
Bunch Grasses
Perennial stream
Capillary fringe
Saturated zone
Confining layer/bedrock

Figure 6-3. General characteristics and functions of a) disturbed and b) undisturbed riparian areas on rangelands. From Elmore (1992). Reproduced with permission from the publisher.

reaches with high incident solar radiation, whereas low amounts of epilithic diatoms and blue-green algae dominated in shaded reaches. Periphyton biomass was found to be significantly correlated with incident solar radiation.

While densities of macroinvertebrates in forested streams typically increase in response to increased periphyton production, the effect of stimulated algal growth in rangeland streams is less clear. Tait et al. (1994) found that biomass, but not density, of macroinvertebrates was greater in reaches with greater periphyton biomass. The higher biomass was a consequence of large numbers of *Dicosmoecus* larvae, a large-cased caddisfly that can exploit filamentous algae. Consequently, any potential benefits of increased invertebrate biomass to organisms at higher trophic levels, including salmonids, may be minimal, because these larvae are well protected from fish predation by their cases. Tait et al. (1994) suggest that these organisms may act as a trophic shunt that prevents energy from being transferred to higher trophic levels.

Evidence of negative effects of livestock grazing on salmonid populations is largely circumstantial, but *is* convincing nonetheless. Platts (1991) found that in 20 of 21 studies identified, stream and riparian habitats were degraded by livestock grazing, and habitat improved when grazing was prohibited in the riparian zone. Fifteen of the 21 studies associated decreasing fish populations with grazing. Although they caution that some of these studies may be biased because of a lack of pregrazing data, the negative effects of grazing on salmonids seem well supported. Storch (1979) reported that in a reach of Camp Creek, Oregon, passing through grazed areas, game fish made up 77% of the population in an enclosure, but only 24% of the population outside the enclosure. Platts (1981) found fish density to be 10.9 times higher in ungrazed or lightly grazed meadows of Horton Creek, Idaho, compared to an adjacent heavily grazed reach. Within an enclosure along the Deschutes River, Oregon, the fish population shifted from predominately dace (*Rhinichthys* sp.) to rainbow trout over a ten-year period without grazing (Claire and Storch 1983). Platts (1991) cites other examples of improved habitat conditions resulting in increased salmonid populations.

6.3 Agriculture

Approximately 12% of the total land area in Washington, Oregon, and Idaho is dry cropland, with an additional 4% devoted to irrigated agriculture. Wheat, barley, and hay account for approximately 44% of the total harvested cropland, with fruits, nuts, berries, hops, peppermint, dry peas, and grass seed all contributing significantly to the total acreage (Jackson 1993). Like the other forms of food and fiber production, farming results in massive alterations of the landscape and the aquatic and riparian ecosystems contained therein. In general, the effects of agriculture on the land surface are more severe than logging or grazing because vegetation removal is permanent and disturbances to soil often occur several times per year. In addition, much agriculture takes place on the historical floodplains of river systems, where it has a direct impact on stream channels and riparian functions. Furthermore, irrigated agriculture frequently requires diversion of surface waters, which decreases water availability and quality for salmonids and other aquatic species (see Section 6.8). Qualitative summaries of the historical effects of agriculture on aquatic ecosystems have been reported by Smith (1971), Cross and Collins (1975), Gammon (1977), and Menzel et al. (1984).

6.3.1 Effects on Vegetation

In the Pacific Northwest, natural grasslands, woodlands, and wetlands have been eliminated to produce domestic crops. For example, in the Willamette Valley of Oregon, the original fire-maintained prairies and floodplain forests were replaced with croplands (Johannessen et al. 1971). Replacement of natural forest and shrubland vegetation with annual crops frequently results in large areas of tilled soil that become increasingly compacted by machinery and are only covered with vegetation for a portion of the year. Commonly, little or no riparian vegetation is retained along streams as farmers attempt to maximize acreage in production. While there is potential to restore agricultural lands to more natural communities, conversions to croplands in most instances have been permanent alterations of the landscape.

6.3.2 Effects on Soils

Agriculture involves repeated tillage, fertilization, pesticide application, and harvesting of the cropped acreage. The repeated mechanical mixing, aeration, and introduction of fertilizers or pesticides significantly alters physical soil characteristics and soil microorganisms. Further, tillage renders a relatively uniform characteristic to soils in the cropped areas. Although tillage aerates the upper soil, compaction of fine textured soils typically occurs just below the depth of tillage, altering the infiltration of water to deep aquifers. Other activities requiring farm machinery to traverse the cropped lands, and roads along crop margins, causes further compaction, reducing infiltration and increasing surface runoff. Where wetlands are drained for conversion to agriculture, organic materials typically decompose, significantly altering the character of the soil. In extreme cases, the loss of organic materials results in "deflation," the dramatic lowering of the soil surface. Soil erosion rates are generally greater from croplands than from other land uses (see Section 6.3.4), but vary with soil type and slope.

6.3.3 Effects on Hydrology

Changes in soils and vegetation on agricultural lands typically result in lower infiltration rates, which yield greater and more rapid runoff. For example, Auten (1933) suggested that forested land may absorb fifty times more water than agricultural areas. Loss of vegetation and soil compaction increase runoff, peak flows, and flooding during wet seasons (Hornbeck et al. 1970). Reduced infiltration and the rapid routing of water from croplands may also lower the water table, resulting in lower summer base flows, higher water temperatures, and fewer permanent streams. Typically, springs, seeps, and headwater streams dry up and disappear, especially when wetlands are ditched and drained.

Water that is removed from streams and spread on the land for irrigated agriculture reduces streamflows, lowers water tables, and leaves less water for fish. Often the water is returned considerable distances from where it was withdrawn, and the return flows typically raise salinity and temperature in receiving streams. Extreme examples of this occur in many rivers east of the Cascades and in the Central Valley of California. The flows of these rivers are naturally low in late summer, but the additional losses from irrigation accentuate low flows. Reductions in summer base flows greatly degrade water quality because the water warms more than normal and causes increased evaporation, which concentrates dissolved chemicals and increases the respiration rates of aquatic life.

Streams are typically channelized in agriculture areas, primarily to reduce flood duration and to alter geometry of cropped lands to improve efficiency of farm machinery. Because peak flows pass through a channelized river system more quickly, downstream flood hazards are increased (Henegar and Harmon 1971). When channelization is accompanied by widespread devegetation, the severity of flooding is increased, such as occurred in the Mississippi Valley in 1993. On the other hand, channelization of streams leads to decreases in summer base flows because of reduced groundwater storage (Wyrick 1968), which can limit habitat availability for fish and increase crowding and competition. In more extreme cases, streams may dry completely during droughts (Gorman and Karr 1978; Griswold et al. 1978).

6.3.4 Effects on Sediment Transport

Because of the intensity of land use, agricultural lands contribute substantial quantities of sediment to streams. The Soil Conservation Service (SCS 1984) estimated that 92% of the total sediment yield in the Snake and Walla Walla River basins of southeastern Washington resulted from sheet and rill erosion from croplands—lands that accounted for only 43% of the total land area. The loss of vegetative cover increases soil erosion because raindrops are free to detach soil particles (splash erosion). Fine sediments mobilized by splash erosion fill soil interstices, which reduces infiltration, increases overland flow, and facilitates sheet and rill erosion. Agricultural practices typically smooth and loosen the land surface, enhancing the opportunity for surface erosion. When crop lands are left fallow between cropping seasons, excessive erosion can greatly increase sediment delivery to streams (SCS 1984). Mass failures are probably rare on most agricultural lands because slopes are generally gentle; however, sloughing of channel banks may occur in riparian zones in response to vegetation removal.

6.3.5 Effects on Thermal Energy Transfer and Stream Temperature

Removal of riparian forests and shrubs for agriculture reduces shading and increases wind speeds, which can greatly increase water temperatures in streams passing through agricultural lands. In addition, bare soils may retain greater heat energy than vegetated soils, thus increasing conductive transfer of heat to water that infiltrates the soil or flows overland into streams. In areas of irrigated agriculture, temperatures increases during the summer are exacerbated by heated return flows (Dauble 1994). These effects are discussed in greater detail on Section 6.1.5.

6.3.6 Effects on Nutrient and Solute Transport

Agricultural practices may substantially modify the water quality of streams. Omernik (1977), in a nationwide analysis of 928 catchments, demonstrated that streams draining agricultural areas had mean concentrations of total phosphorus and total nitrogen 900% greater than those in streams draining forested lands. Smart et al. (1985) found that water quality of Ozark streams was more strongly related to land use than to geology or soil. Exponential increases in chlorine, nitrogen, sodium, phosphorus, and chlorophyll-a occurred with increases in percent pasture in streams draining both forested and pastured catchments, and fundamental alterations in chemical habitat resulted as the dominant land use changed from forest to pasture to urban. Stimulation of algal growth by nutrient enrichment from agricultural runoff may affect other aspects of water quality. As algal blooms die off, oxygen consumption by microbial organisms is increased and can substantially lower total dissolved oxygen concentrations in surface waters (Waldichuk 1993). Nutrient enrichment from agricultural runoff has been found to significantly affect water quality in two rivers in interior British Columbia. Die-off of nutrient-induced algal blooms resulted in significant oxygen depletion (concentrations as low as 1.1 $mg \cdot L^{-1}$) in the Serpentine and Nicomekl rivers during the summer, which in turn caused substantial mortality of coho salmon.

6.3.7 Effects of Fertilizer and Pesticide Use

The two most commonly used agricultural chemicals, herbicides and nitrogen, are frequently found in groundwater in agricultural areas. McBride et al. (1988) report that atrazine is the herbicide most often detected in corn belt groundwater. In Oregon, groundwater nitrogen concentrations at or above health advisory levels were found in Clatsop, Marion, Deschutes, Morrow, Umatilla, Union, and Malheur counties, and elevated levels were reported for Multnomah, Linn, and Lane counties (Vomocil and Hart 1993). Because of the lack of a statistically representative sample of groundwater in the region's agricultural areas, the degree and extent of contamination is unknown.

Unlike native vegetation, agricultural crops require substantial inputs of water, fertilizer, and biocides to thrive. Currently used pesticides, although not as persistent as previously-used chlorinated hydrocarbons, are still toxic to aquatic life. Where biocides are applied at recommended concentrations and rates, and where there is a sufficient riparian buffer, the toxic effects to aquatic life may be minimal. However, agricultural lands are also characterized by poorly-maintained dirt roads and ditches that, along with drains, route sediments, nutrients, and biocides directly into surface waters. Thus, roads, ditches, and drains have replaced headwater streams; but rather than filter and process pollutants, these constructed systems deliver them directly to surface waters (Larimore and Smith 1963).

6.3.8 Effects on Physical Habitat Structure

Agricultural practices typically include stream channelization, large woody debris removal, construction of revetments (bank armoring), and removal of natural riparian vegetation. Each of these activities reduces physical habitat complexity, decreases channel stability, and alters the food base of the stream (Karr and Schlosser 1978). Natural channels in easily eroded soils tend to be braided and meander, creating considerable channel complexity as well as accumulations of fallen trees. Large wood helps create large, deep, relatively permanent pools (Hickman 1975), and meander cutoffs; the absence of snags simplifies the channel. Channelization lowers the base level of tributaries, stimulating their erosion (Nunnally and Keller 1979). The channelized reach becomes wider and shallower, unless it is revetted, in which case bed scour occurs that leads to channel downcutting or armoring. Channel downcutting leads to a further cycle of tributary erosion. Richards and Host (1994) reported significant correlations between increased agriculture at the catchment scale and increased stream downcutting. Incised channels in an agricultural region were found to have less woody debris and more deep pools than nonincised channels (Shields et al. 1994).

6.3.9 Effects on Stream Biota

Agricultural practices also cause biological changes in aquatic ecosystems. In two States typified by extensive agricultural development and with extensive statewide ecological stream surveys, instream biological criteria were not met in 85% of the sites (Ohio EPA 1990; Maxted et al. 1994a). Nonpoint sources of nutrients and physical habitat degradation were identified as causes of much of the biological degradation. In another study, Maxted et al. (1994b) also demonstrated that the amount of shading had marked effects on stream temperatures and dissolved oxygen concentrations (e.g., Figure 6-4). In some agricultural stream reaches without riparian vegetation, the extremes exhibited in both temperature and DO would preclude the survival of all but the most tolerant organisms. Higher temperatures increase respiration rates of fish, increasing oxygen demand at the same time that oxygen is depleted by stimulated plant respiration at night. During daylight hours, high plant respiration (elevated by greater nutrient concentrations, higher temperatures, and lower flows) may produce gas supersaturation and cause fish tissue damage. Smith (1971) reported that 34% of native Illinois fish species were extirpated or decimated, chiefly by siltation, and lowering of water tables associated with drainage of lakes and wetlands. Although point sources were described by Karr et al. (1985) as having intensive impacts, nonpoint sources associated with agriculture were considered most responsible for declines or extirpations of 44% and 67% of the fish species from the Maumee and Illinois drainages, respectively. Sixty-three percent of California's native fishes are extinct or declining (Moyle and Williams 1990), with species in agricultural areas being particularly affected. Nationwide, Judy et al. (1984) reported that agriculture adversely affected 43% of all waters and was a major concern in 17%.

Modification of physical habitat structure has been linked with changes in aquatic biota in streams draining agricultural lands. Snags are critical for trapping terrestrial litter that is the primary food source for benthos in small streams (Cummins 1974), and as a substrate for algae and filter feeders in larger rivers. Benke et al. (1985) describe the importance of snags to benthos and fish in rivers with shifting (sand) substrates. Such systems, typical of agricultural lands, support the majority of game fish and their prey. Marzolf (1978) estimates 90% of macroinvertebrate biomass was attached to snags. Hickman (1975) found that snags were associated with 25% higher standing crops for all fish and 51% higher standing crops for catchable fish. Fish biomass was 4.8–9.4 times greater in a stream side with instream cover than in the side that had been cleared of all cover (Angermeier and Karr 1984). Gorman and Karr (1978) reported a correlation of 0.81 between fish species diversity and habitat diversity (substrate, depth, velocity). Shields et al. (1994) found that incised channels in agricultural regions supported smaller fishes and fewer fish species.

On a larger scale, habitat and reach diversity must be great enough to provide refugia for fishes during temperature extremes, droughts, and floods

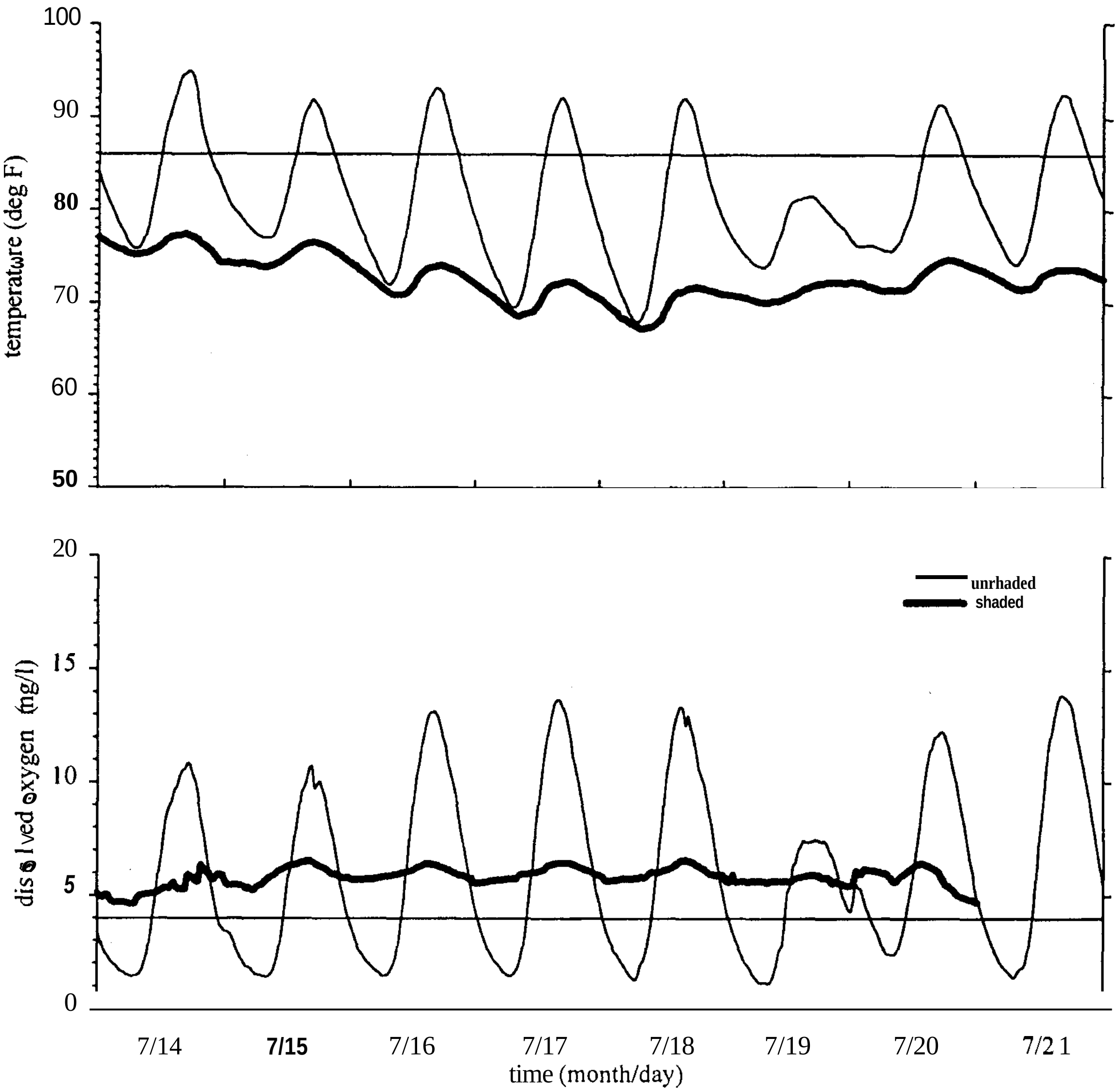


Figure 6-4. Diel fluctuations in temperature (top) and dissolved oxygen (bottom) in shaded and unshaded reaches of Mudstone Branch/Wharton Branch. From Maxted et al. (1994b). Reproduced with permission from the principal author.

(Matthews and Heins 1987). If refugia occur, fishes in agricultural streams can rapidly recolonize disturbed habitats and reaches. However, loss of refugia, alterations in water tables, simplifications of channels, and elimination of natural woody riparian vegetation symptomatic of agricultural regions creates increased instability and results in stream degradation (Karr et al. 1983).

6.4 Urbanization

Urban areas occupy only 2.1% of the Pacific Northwest regional land base (Pease 1993), but the impacts of urbanization on aquatic ecosystems are severe and long-lasting. Future projections suggest that urban areas will occupy an increasing fraction of the landscape. From 1982 to 1987, lands devoted to urban and transportation uses increased by 5.2% (45,346 hectares [123,813 acres]) in the Pacific Northwest. In the Puget Sound area, the population is predicted to increase by 20% between 1987 and the year 2000, requiring a 62% increase in land area developed for intense urbanization (PSWQA 1988). As urban areas continue to expand, natural watershed processes will be substantially altered.

Urbanization has obvious effects on soils and natural vegetation that, in turn, affect hydrologic and erosional processes, as well as physical characteristics of aquatic habitats. Urban developments, including roads, buildings, sidewalks, and other impervious surfaces, greatly reduce water infiltration, which alters the routing and storage of water in the basin. Many of the resulting changes are intended and make the land more amenable to specific human uses (e.g., transportation, human habitation), but other important resource values (e.g., water supplies, fisheries, and wildlife) may be damaged by unintended effects on aquatic ecosystems, including increased peak flows, channel erosion, landslides, pollution, and channelization.

6.4.1 Effects on Vegetation

Urbanization causes severe and essentially permanent alteration of natural vegetation. The total vegetated area in the basin is typically diminished, and replacement vegetation (e.g., lawns, ornamental plants) often requires large quantities of water and fertilizers for growth. In addition, riparian corridors are frequently constricted, disabling or altering riparian processes. The loss of riparian vegetation reduces inputs of large woody debris and smaller organic detritus including leaves. Stream channels and banks are deprived of stability provided by large woody debris and the roots of riparian vegetation.

6.4.2 Effects on Soils

The effects of urbanization on soils can be divided into two phases. During urban construction, significant soil displacement, alteration, and movement occurs associated with grading, filling, and hauling activities. Once land conversion is complete, much of the surface soil is covered with buildings, concrete, and asphalt. In most residential areas, soils may be exposed, but they are generally altered and fertilized to support domesticated vegetation. Because of this dramatic alteration, the ecological functions that occur in the soil are likely greatly diminished, and these changes may be permanent.

6.4.3 Effects on Hydrology

Urbanization significantly influences hydrologic processes, increasing the magnitude and frequency of peak discharges and reducing summer base flows (Klein 1979; Booth 1991). These changes occur primarily because of increases in the impervious surface and the replacement of complex, natural drainage channels with a network of storm pipes and drainage ditches (Lucchetti and Fuerstenberg 1993). In urban areas, infiltration is reduced as 1) soils are stripped of vegetation, compacted, and or paved; 2) internal draining depressions are graded; 3) subsurface flow is intercepted by drains and discharged to streams; and 4) buildings are erected (Booth 1991). Instead of infiltrating into the soil, storm water is quickly delivered to the channel, resulting in a more episodic flow regime with higher peak flows and reduced base flows. In nonurban areas west of the Cascades, rainfall intensities are lower than the rate of infiltration, and subsurface flows predominate (Dunne et al. 1975). Only a small portion of the watershed contributes overland runoff; the remaining water infiltrates and becomes part of the subsurface regime. In arid and semi-arid eastside systems, overland runoff is more common because of higher rainfall intensities, sparse vegetation, and shallow, less permeable soils. Runoff generally travels quickly from the hillslopes to the channel, and virtually all parts of the watershed contribute to storm runoff. Although eastside runoff is primarily overland flow, urbanization increases the efficiency of water delivery to the channel. Culverts and drainpipes are straighter and provide a more direct and more efficient flow to the stream channel.

Increases in storm runoff caused by decreased infiltration also may result in more frequent flood events (Klein 1979). Using a model that incorporated historical storm data for Hylebos Creek, Washington, Booth (1991) found that over a 40-year simulation period, storm flows from an urban area were significantly greater than those from a forested basin. For the fully forested basin, seven floods exceeding the magnitude of a 5-year event were simulated for the 40-year period. In contrast, in the urbanized basin, simulated floods equaled or exceeded the discharge of a 5-year flood event in 39 of the 40 years (Figure 6-5).

Water withdrawals for water supply, industry and food processing can alter the flow regimes and quantity and quality of stream water. Muckleston (1993) reported that public water supplies accounted for 42% and 84%, respectively, of the total withdrawals from surface waters in the Willamette Basin, Oregon, and Puget Sound, Washington; these areas have the highest population densities found in these two States. In the lower Columbia sub-basin, public water supply and industrial usage make up over 80% of total withdrawals. East of the Cascade crest, food processing is generally the most significant industrial use of water though refining primary metals is important locally in the Clark Fork, Kootenai, Spokane, and mid-Columbia sub-basins. The need for water supplies, dependable power, and flood control has led to numerous impoundments on the major Northwest river systems. These reservoirs have altered the natural flow regimes and fish habitats. For example, flows in the Willamette River, which historically reflected annual precipitation patterns, have been substantially altered to accommodate urban water needs. On average, summer low flows are higher than in predevelopment periods because water is now stored during the wet season and released during the summer.

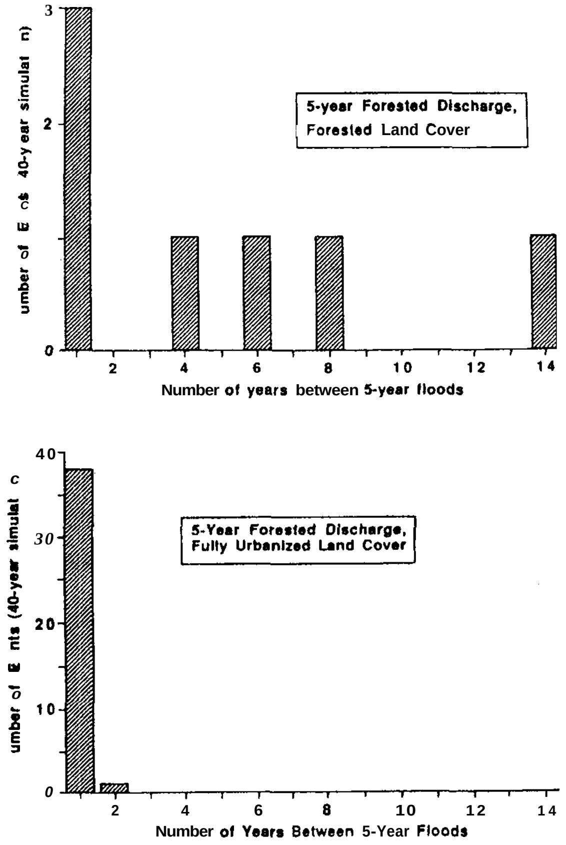


Figure 6-5. Hydrologic Simulation Program Fortran simulation of the Hylebos Creek basin in southwest King County, Washington, under fully forested land cover (top) and fully urbanized condition (bottom) (approximately 40% effective impervious area). Bars show the number of years separating discharge events of 5-year recurrence or greater. For forested condition, the average separation is 5 years (40 years of simulation, 8 events), but the actual spacing varies from 1 year (i.e., successive years) to 14 years. For urbanized condition, discharges exceeding 5-year forested event occur every year except one. Figure reprinted with permission from Booth (1991). ©1991 by the Institute for Environmental Studies, University of Washington.

6.4.4 Effects on Sediment Transport

Loss of vegetation and alteration of soil structure during construction of buildings and roads may increase sediment loading to streams by several orders of magnitude (Klein 1979); however, the effect is likely to be of short duration. Once building and landscaping is complete, surface erosion is reduced, possibly to levels lower than prior to construction because much of the land surface is under impervious surfaces. Specific effects are likely to vary with degree of urbanization, and whether drainage ditches are composed of erodible materials or concrete. Street sweeping and runoff from city streets transports some sediment to storm sewers and ultimately to streams, but the impact of that sediment is negligible. However, contaminants associated with such sediments can have significant impacts on water quality (see Section 6.4.7).

6.4.5 Effects on Thermal Energy Transfer and Stream Temperatures

Changes in riparian vegetation along urban streams can alter the degree of shading provided to the stream, which in turn influences seasonal and diurnal temperature ranges (see Section 3.7). As with other land uses, effects are likely to be greatest for smaller streams that previously had closed canopies. Published examples of changes in temperature regimes caused by urbanization are scarce; however, likely effects are increased maximum temperatures (Klein 1979), greater diel fluctuations, and reduced winter temperatures. Pluhowski (1970 in Klein 1979) found that winter stream temperatures in urban areas were 1.5–3°C lower than in nonurbanized streams on Long Island, New York. Although other land-use activities alter stream temperatures, in urban areas the loss of riparian function is long lasting.

Stream temperatures may also be indirectly affected by changes in hydrology, channel morphology, and the urban microclimate. Klein (1979) suggested that reductions in groundwater inflow may alter natural thermal regimes, resulting in lower winter minimum temperatures and higher summer maximum temperatures. Widening and shallowing of channels caused by greater peak discharges can also influence the rate of energy transfer to and from streams. Air temperatures in urban areas also tend to be warmer than those in surrounding rural areas, which may affect convective and evaporative energy exchange.

6.4.6 Effects on Nutrients and Other Solutes

The primary changes in nutrient cycling are the type and quantity of materials delivered to the stream channel. Large woody debris and leafy detritus are replaced in importance by nutrient loading from sewage and other sources. Novitzki (1973) reported that effluent from a sewage treatment plant in a small town in Wisconsin significantly degraded brook trout habitat downstream of the release point. High nutrient levels from the effluent generally stimulated primary and secondary production; however, under conditions of high temperature and low flow during the summer, heavy oxygen demand from the aquatic vegetation and effluent created critically low dissolved oxygen levels that resulted in fish kills. Omernik (1977) determined that total nitrogen exports from urban areas were second only to agriculturally influenced watersheds.

6.4.7 Effects of Chemical Use

Runoff from the urban areas contains many different types of pollutants depending on the source and nature of activities in the area. Wanielista (1978) identified numerous types of urban nonpoint source pollution including heavy metals, nutrients (phosphates and nitrates), pesticides, bacteria, organics (oil, grease) and dust/dirt. Heavy metal concentrations found in street runoff were 10–100 times greater than treated wastewater effluent, Contributions of grease and oil ranged from 32.8 lb·curb $\text{mile}^{-1}\cdot\text{day}^{-1}$ for industrial areas to 4.9 lb·curb mile-'.day-' for commercial areas (Pitt and Amy 1973). Residential areas fell between (18.6 lb·curb $\text{mile}^{-1}\cdot\text{day}^{-1}$). Klein (1979) reported that 9% of persons that changed their own engine oil in their cars and disposed of used oil by pouring it into storm drains or gutters. In suburban areas, fertilizers, herbicides, pesticides and animal waste are added to the effluent. For example, Bryan (1972) found that pesticide loadings in runoff from urban areas was three-to-four times greater than for rural areas. In industrial areas, runoff may include heavy metals, polychlorinated biphenyls (PCBs), polycyclic aromatic hydrocarbons (PAHs), high Ph concrete dust, and other toxic chemicals (Birch et al. 1992). Water quality degrades as a consequence of these pollutants entering streams, lakes, and estuaries. Biological oxygen demand is increased with the addition of organic materials, and lethal or sublethal effects may occur with influxes of heavy metals, pesticides, PCBs, and PAHs (see Sections 5.1.2 and 6.4.9).

6.4.8 Effects on Physical Habitat Structure

Urbanization frequently results in gross modification of stream and river channels through road construction, the filling of wetlands, encroachment on riparian areas and floodplains, relocation of channels, and construction and maintenance of ditches, dikes, and levees. Urban-related development can influence instream channel structure in a variety of ways. High densities of roads require road crossings, culverts, and other structures that constrain channels and may impede fish migration. Channels are frequently straightened in an attempt to route water quickly through the system and avert flood damage. Rip-rap, concrete, and other forms of channel revetment are commonly employed to counteract the increased erosive force associated with higher discharge volumes. In addition, with increased magnitude and frequency of floods in urban streams and rivers, greater within-stream bedload transport occurs, and channels become less stable (Bryan 1972; Scott et al. 1986). The rates of disturbance from flood events may accelerate to a point that the stream cannot recover between disturbance events. Lucchetti and Fuerstenberg (1992) noted that urbanized streams take on a clean "washed-out" look as channel complexity is lost. Such stream beds are uniform, with few pools or developed riffles, and have substrates dominated by coarser fractions rather than gravel, sand, and silt. The lack of large woody debris inputs exacerbates channel simplification (Lucchetti and Fuerstenberg 1992), causing increased bed scour and fill and changing channel hydraulics at a given maximum flow. These highly modified channels generally provide poor habitat for fish.

In unconstrained urban streams, stream channels may become substantially wider and shallower than streams in rural areas because of higher stream energy and increased erosion of streambanks (Klein 1979). In other areas, streambed morphology is further modified by channel incision, which leaves exposed, near-vertical channel banks (Lucchetti and Fuerstenberg 1992). In areas near the ocean, this can effectively isolate the estuaries from the surrounding riparian zone and essentially create a noninteracting conduit between upriver areas and the sea, Important interactions between the stream and surrounding floodplain are lost.

6.4.9 Effects on Stream Biota

The structure of the biological community and the abundance of aquatic organisms are greatly altered by urban impacts on channel characteristics and water quality. Research indicates that stream quality impairment is correlated to the percentage of watershed imperviousness. Impaired water quality becomes noticeable at 8%-12% imperviousness and becomes severe above 30% imperviousness (Klein 1979; Pedersen and Perkins 1986; Limburg and Schmidt 1990). In a study of northern Virginia streams, Jones and Clark (1987) found that the taxonomic composition of macroinvertebrates was shifted markedly by urbanization, though development had minor effect on the total insect densities. Relative abundance of Diptera (primarily chironomids) increased at the most developed sites, and more sensitive orders, including Ephemeroptera (mayflies), Coleoptera (beetles), Megaloptera (dobsonflies), and Plecoptera (stoneflies), decreased. The response of Tricoptera (caddisflies) was variable. Pedersen and Perkins (1986) showed that a rural stream had twice the functional diversity of an urban steam. Those organisms that persisted were adapted to extreme bed instability.

Fish are also adversely affected by urbanization. Limburg and Schmidt (1990) demonstrated a measurable decrease in spawning success of anadromous species (primarily alewives) for Hudson River tributaries from streams with 15% or more of the watershed area in urban land use. In Kelsey Creek, Washington, urban development resulted in a restructuring of the fish assemblage in response to habitat degradation (Bryan 1972; Scott et al. 1986). Coho salmon appeared to be more sensitive than resident cutthroat trout to habitat alteration, increased nutrient loading, and degradation of the intragravel environment in the stream. In a study of Puget Sound streams, Lucchetti and Fuerstenberg (1992) found that fish assemblages in small urbanized streams have been dramatically altered or lost. They concluded that coho are of particular concern in urbanized areas because of their specific habitat needs (smaller streams, relatively low velocity niches, and especially large pools). Their study found that as impervious surfaces increased fish species diversity and coho abundance declined and resident cutthroat trout dominated.

Recent studies in the Pacific Northwest suggest that pollution from urban areas may be having insidious effects on anadromous salmonids. Arkoosh et al. (1991) found that juvenile chinook salmon that migrate through an urban estuary contaminated withPCBs and PAHs bioaccumulated these pollutants and exhibited a suppressed immune response compared to fish from an uncontaminated rural estuary. In subsequent studies, Arkoosh et al. (1994) exposed juvenile salmon collected from the same two estuaries, as well as their respective releasing hatcheries, to the pathogen *Vibrio anguillarum*. Salmon from the urban estuary exhibited higher mortality rate after 7 days than unexposed fish from the releasing hatchery. In contrast, no difference in mortality rates from this pathogen were observed between the salmon from the uncontaminated estuary and its releasing hatchery. Casillas et al. (1993) found that juvenile chinook exposed to PAHs and PCBs in an urban estuary showed suppressed immune competence and suppressed growth for up to 90 days after exposure, while juvenile chinook from a nonurban estuary did not develop these symptoms. They suggested that suppressed immune function, reduced survival, and impaired growth, result from increased chemical-contaminant exposure of juvenile chinook as they move through urban estuaries on their way to the ocean. The role of contaminants in the overall decline of salmonids is not known; however, these studies indicate that contaminant exposure is perhaps an overlooked cause of mortality for populations that migrate through urbanized streams, particularly because exposure occurs during the physiologically stressful period of smoltification.

6.5 Sand and Gravel Mining

Gravel and sand removal from streams and adjacent floodplains is common in many areas of the Pacific Northwest, particularly near and in low-gradient reaches of rivers west of the Cascade range. In Oregon, permits are required for removal of gravel or sand in excess of 38.3 m^3 (50 yr^3) (OWRRI 1995). Since 1967, the Oregon Division of State Lands has issued over 4,000 permits for gravel removal (OWRRI 1995), and between 1987 and 1989, a total of 1767 dredge, fill, and aggregate extraction permits were processed, 718 of which were new permits (Kaczynski and Palmisano et al. 1993a). Because there are no permit requirements for gravel extraction of less than 38.3 m^3, little information exists regarding the extent of small-scale gravel mining in Oregon. In Washington, large amounts of gravel are associated with glacial deposits and, thus, instream mining has decreased in recent years as extraction has shifted towards glacial and floodplain deposits (Dave Norman, Washington Department of Natural Resources, personal communication). Nevertheless, mining activity occurs near or in most major rivers west of the Cascade Range (Palmisano *et* al. 1993a, 1993b). Sandecki

(1989) reported that production of sand and gravel in California during 1986 exceeded 128 million short tons. The greatest demand for gravel and sand is associated with industrial development, and because of the expense of transporting gravel, mining is most prevalent around urban areas, along highways, or near other major construction sites. Most gravel permit sites in Washington are located near or in urban areas and along the Interstate 5 corridor (Figure 6-6). In Oregon, gravel production has generally risen between 1940 and 1990; however, gravel mining activity peaked during the 1960s and early 1970s with construction of the John Day, Green Peter, and Foster dams (OWRRI 1995). The majority of gravel mining in Oregon occurs in the Willamette Valley.

Two recent reviews focused on effects of gravel removal on hydrology and channel morphology (Sandecki 1989; Collins and Dunne 1990), and a third focused on effects on salmonids in Oregon (OWRRI 1995). Much of the material contained in this section comes from these three sources.

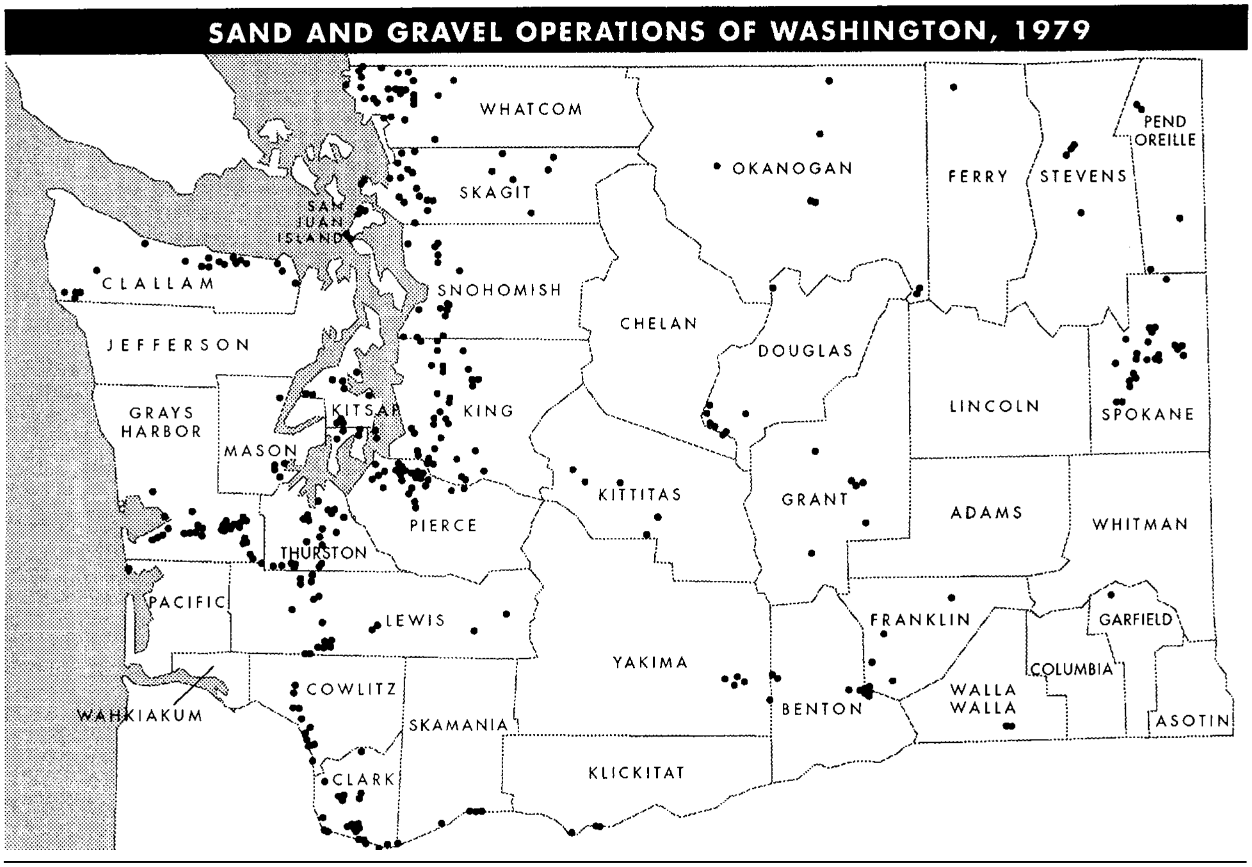

SAND AND GRAVEL OPERATIONS OF WASHINGTON, 1979
WHATCOM
OKANOGAN
FERRY
STEVENS
PEND OREILLE
SKAGIT
SAN JUAN ISLAND
CLALLAM
SNOHOMISH
CHELAN
JEFFERSON
DOUGLAS
KITSAP
KING
GRAYS HARBOR
MASON
LINCOLN
SPOKANE
KITTITAS
GRANT
PIERCE
THURSTON
ADAMS
WHITMAN
PACIFIC
LEWIS
FRANKLIN
GARFIELD
YAKIMA
COLUMBIA
COWLITZ
WAHKIAKUM
BENTON
WALLA WALLA
ASOTIN
SKAMANIA
CLARK
KLICKITAT

Figure 6-6. Sand and gravel operations of Washington, 1979. From Palmisano et al. (1993a). Reproduced with permission from the author.

6.5.I Effects on Geomorphology and Sediment Transport

Removal of sand and gravel from within a stream channel may fundamentally alter the way in which water and sediment are carried through a system, resulting in altered channel morphology, decreased stability, accelerated erosion, and changes in the composition and structure of the substrate (Sandecki 1989; OWRRI 1995). The extent of effects depends on many site-specific characteristics, including the geomorphic setting (e.g., stream gradient and nature of bed material), the quantity of material extracted relative to the sediment supply, and the hydrologic and hydraulic conditions within the stream reach.

The effects of gravel mining on the stream environment involve a complex interplay between direct effects of channel modification and altered substrate composition, and the resulting alteration of erosional and depositional processes, which in turn feed back to cause further changes in channel configuration. Excavation of materials from the stream bed results in immediate changes to channel morphology. Newly created mining pits within streams are highly unstable and tend to migrate up or downstream in response to scouring and deposition of sediments (Lee et al. 1993). Thus, the physical effects of mining pits propagate away from the immediate excavation site (Sandecki 1989; OWRRI 1995). In undisturbed stream channels, coarser materials have a tendency, through hydraulic sorting, to "armor" the stream bed, increasing its resistance to scour (Lagasse et al. 1980). Finer materials work their way into deeper layers. Gravel mining disrupts the armor layer, leaving smaller materials at the bed surface that are more easily mobilized by streamflow; thus bedload movement occurs at lower stream velocities following gravel mining (Sandecki 1989).

Removal of bed material and increased bedload transport can combine to cause downcutting of the stream channel in both upstream and downstream directions (Sandecki 1989; OWRRI 1995). Downstream progression may result from reduced bed material discharge or decreased size of bed material, while upstream progression occurs when gravel extraction increases the river gradient (OWRRI 1995). In some cases, downcutting may occur until sand, gravel, and cobble are completely removed and underlying bedrock is exposed Downcutting may cause streambanks to collapse, introducing additional sediments into the stream (Collins and Dunne 1990).

Collins and Dunne (1990) recently reviewed case histories on the effects of gravel extraction on downcutting and found several examples where stream channels lowered 4 to 6 m in response to gravel mining (Table 6-6). In several cases, downcutting occurred over several decades; however, in one instance, a drop in bed elevation of 4.5 m occurred during two flood events that spilled into a large mining pit in Tujunga Wash, California, demonstrating that downcutting can occur rapidly under extreme circumstances. Kondolf and Swanson (1993) reported that gravel extraction below a dam in a Sacramento River tributary resulted in downcutting of more than 5 m and caused a shift from a highly braided channel to a single channel. Downcutting was severe in part because the dam prevented recruitment of gravels from upstream areas; however, reduced peak flows may have compensated for reduced sediment recruitment by reducing scouring. This example highlights the fact that effects of gravel mining depend on the cumulative effects of other activities in the watershed.

While the effects of off-channel mining are likely to be less direct, they may nevertheless be significant. Frequently, berms, dikes, or revetments are constructed to prevent flood flows from spilling into the excavation area and to reduce bank erosion. These structures prevent lateral migration of the stream channel, which may be important in recruiting gravels from streambanks. During high flows, water is constrained to a narrow channel, which increases the velocity and, hence, the erosive potential of the discharge. Artificially constricted channels, like excavations, may thus lead to degradation of the stream bed. Bar scalping also may affect erosion processes. When bar height is reduced, instream bars may be more prone to erosion when water level rises again (Collins and Dunne 1990).

In summary, the effects of gravel extraction on stream channels may include local adjustments, increased meandering or widening of the stream channel, changes in thalweg configuration, altered pool-riffle sequences, shifts from braided to single-thread channels, and downcutting of the channel bed (Sandecki 1989). Gravel mining may also change the frequency and extent of bedload movements and increase the amount of suspended fine sediments and turbidity in the water column. Turbidity caused by excavation generally decreases shortly after mining activity ceases; however, turbidity caused by changes in erosion potential may persist until the streambed restabilizes (reviewed in OWRRI 1995). Fine sediments may settle in gravel pits or travel downstream to settle in other slow-water areas. As a result, downstream substrates may be covered with sand, mud, and silt.

6.5.2 Effects on Hydrology

Gravel mining likely has little effect on the total amount of water moving through a stream system; however, it may significantly affect the routing and timing of streamflow. Both downcutting and channel simplification increase the hydraulic efficiency of the stream—water is routed more quickly through the system, especially during periods of high flow (Sandecki 1989). This increased efficiency may reduce the probability of overbank flooding (Collins and Dunne 1990). The elimination of overbank flows prevents the recharge and subsequent release of water from the floodplain, which in turn results in flashier streamflows. In addition, channel downcutting may drain shallow groundwater, and lower the water table (Sandecki 1989; Collins and Dunne 1990; OWRRI 1995). Loss of shallow groundwater storage reduces summer base flows and may also lead to loss of riparian vegetation (Sandecki 1989).

Table 6-6. Case histories relating the effects of gravel extraction on channel morphology and hydrology of streams in Washington, Oregon, and California.

Location	Activitv	Effects
WASHINGTON		
Humptulips, Wynoochee, and Satsop Rivers[*]	Gravel bar scalping	Minimum rates of gravel extraction exceeded replenishment rate. Channel degradation (lowering) occurred at some sites.
White River'	Gravel extraction (partly for flood prevention), diking, and straightening	Aggradation in lower reaches, degradation in upper reaches.
Skykomish River[†]	In-channel gravel mining	Diminished size of gravel bars that were mined, as well as upstream and downstream sites. Reduced rate of bank erosion.
CALIFORNIA		
Cache Creek[†]	In-channel extraction during dry season	Channel degradation up to 9 m (avg 5 m) over 21-year period. Increased flood capacity has eliminated overbank flooding and is preventing soils from being deposited on flood plain. Drop in ground water table has shifted system from a "drain" system to recharge system. Loss of aquifer storage potential.
Russian River[†]	Gravel extraction	Channel degradation up to 6 m (avg 4 m). Exposure of bedrock substrate.
Dry Creek[†]	Gravel extraction	Channel degradation up to 4 m. Riparian vegetation has died, probably in response to lowering water table.
Tujunga Wash[†, ‡]	Off-channel gravel mining	Gravel pit was inundated by 1969 floods. Headward scour up to 4.5 m extended 790–914 m upchannel.
Redwood Creek[†]	Channelization, levee construction, gravel mining to low water level	Alternating lowering of bed by mining and raising of bed from redeposition. Shift in thalweg. Gravel bars removed annually by mining contributing to channel destabilization. Headward degradation of channel.
Stony Creek[§]	In-channel gravel mining	Channel shifted from braided configuration to single, incised, meandering channel. Degradation up to 5 m. Obliteration of natural low-flow channels. Effects modified by changes in flow regime due to construction of dam upstream.
OREGON		
Willamette River[¶, #]	Sand and gravel extraction	Channel degradation of approximately 0.3 m per year over 20- to 30-year period. Degradation because of combined effects of sand and gravel extraction, bank stabilization, dams, watershed changes and natural geological events.

[*] Collins and Dunne (1989).
[†] Collins and Dunne (1990).
[‡] Scott (1973).
[§] Kondolf and Swanson (1993).
[¶] Klingeman (1993).
[#] OWRRI (1995).

6.5.3 Effects on Thermal Energy Transfer and Stream Temperature

The most likely changes in heat transfer processes resulting from gravel mining are increased heat exchange from the loss of riparian vegetation and alteration of the surface-to-volume ratio of the stream (OWRRI 1995). As discussed in Section 3.7, heat exchange is greater in wide, shallow streams than in narrow deep channels, so temperatures may increase or decrease depending on the specific change in channel morphology that follows gravel extraction. Stream temperatures may also increase because of inputs of heated water from off-channel ponds created by excavation (OWRRI 1995).

6.5.4 Effects on Nutrients and Other Solutes

We found no published information regarding the effects of instream gravel mining on nutrient cycling or availability. However, if the water table in the floodplain is lowered, floodplain soils may shift from reducing environments to oxidizing environments. Because the form of nitrogen and other solutes depends on the redox potential of the subsurface environment (Section 6.2.6) the availability of nitrate nitrogen and other solutes may increase in response to the oxidizing environment. Nutrient inputs are also affected where riparian vegetation is modified or eliminated.

6.5.5 Effects on Physical Habitat Structure

Most concern regarding the effects of gravel and sand mining on salmonids has focused on spawning habitats. Extraction of gravels may directly eliminate the amount of gravels available for spawning if the extraction rate exceeds the deposition rate of new gravels in the system. The areal extent of suitable spawning gravels may be reduced where degradation reduces gravel depth or exposes bedrock. In addition, decreases in the stability of streambeds can potentially increase embryo and alevin mortality because of scouring of gravel beds. Deposition of fine sediments downstream of mining activities may reduce the quality of these areas as spawning habitats (Kondolf 1994).

Widening and shallowing of stream channels in response to gravel mining may affect the suitability of stream reaches as rearing habitat for juveniles, particularly during summer low-flow periods, when deeper waters are important for survival. Similarly, a reduction in pool frequency may adversely affect migrating adults that require holding pools during their upstream migrations,

6.5.6 Effects on Stream Biota

Gravel mining can change the abundance and composition of species at lower trophic levels. Increased turbidity reduces light penetration, thereby affecting the production of benthic algae (OWRRI 1995). Aquatic invertebrates, which are an important prey for stream-dwelling salmonids, can be disrupted by disturbance of the substrate during mining (AFS 1988) or by changing substrate composition or covering of substrate with fine sediments (Hicks et al. 1991a). Potential effects on invertebrates include changes in species composition, reduced biomass, and slowed biotic colonization (OWRRI 1995).

OWRRI (1995) found few studies that address other effects of gravel mining on salmonids; however, qualitative inferences can be drawn from studies of the effects of logging, grazing, and other activities where physical changes are comparable. Salmonids require clean, well-oxygenated waters for successful incubation of embryos and alevins. Mechanical disturbance of spawning beds by mining equipment can potentially lead to high mortality rates of embryos and alevins. The OWRRI (1995) report cites one study where angler wading caused high mortality (43%–96%)of alevins with only one to two passes per day. It is likely that gravel mining equipment would be substantially more damaging to incubating embryos and alevins than anglers.

Turbidity reduces the reactive distance of fish during foraging (Barrett et al. 1992), clogs or damages buccal or gill membranes, and inhibits normal activities (Hicks et al. 1991a; Barrett et al. 1992). Sigler et al. (1984) reported that turbidities ranging from 25–50 NTU (nephelometric turbidity units) reduced growth and increased the tendency of young coho salmon and steelhead trout to emigrate from laboratory streams. Other direct effects of turbidity on fish are discussed in greater detail in Section 5.1.2. Potential effects on fish assemblages include reduced salmonid production, reductions in total biomass, decreased species diversity, and shifts away from species preferring clear waters towards species that are tolerant of high turbidities (OWRRI 1995). Those species that are most susceptible to increased fine sediments are those that rely heavily on benthic organisms for food or clean gravels for spawning, such as salmon and trout (OWRRI 1995).

6.6 Mineral Mining

In the Pacific Northwest and California, mining has had substantial influence on environmental conditions and patterns of human settlement. Mining provided the initial driving force for the ecological transformation of portions of the interior Northwest.

The discovery of gold in California and the western interior region in the 1860s catalyzed the large influx of people intent on extracting minerals from streams and mountainous slopes. This provided a wedge into the interior-montane ecosystems from the coastal regions for the cultural transformation of the Pacific northwest. Mining as practiced in the 1800s was especially disruptive to stream ecosystems, Hydraulic mining sluiced hillslopes down into streams, causing siltation of waterways and degradation of riparian habitats. Extensive cutting of inland forests was undertaken to provide trusses for mine tunnels and wooden viaducts, sluices, and flumes. By 1870, cattle and sheep that had been brought in to feed miners grazed throughout the intermontane Northwest (Robbins and Wolf 1994). Hydraulic mining of the main river valleys of California's Salmon River from 1870 to 1950 is estimated to have produced about 12.1 million m^3 (15.8 million yd^3) of sediments (PFMC Habitat Committee 1994). The effects there and elsewhere are still being felt today as sediments and pollutants derived from mine tailings continue to enter streams. The PSMFC (1994) reports that mining is responsible for polluting 19,350 km (12,000 miles) of rivers and streams in the western United States. Recovery rates of degraded streams vary, ranging from 20 years for areas with no acid drainage to generations for coal mines (with acid drainage), and radioactive phosphate and uranium mines (AFS 1988). Before the Surface Mining Control and Reclamation Act of 1977, little thought was given by many to post-mining effects and reclamation efforts (Nelson et al. 1991). However, some States (e.g., Oregon) have enacted more stringent laws regulating certain types of mineral extraction (Field 1993).

Minerals are extracted by several methods that can be combined into two broad categories. Surface mining includes dredging, hydraulic mining, strip mining, and pit mining. Underground mining utilizes tunnels or shafts to extract minerals by physical or chemical means. Surface mining probably has greater potential to affect aquatic ecosystems, although pollution associated with all forms of mining activities may be damaging to aquatic life. Specific effects on aquatic systems depend on the extraction and processing methods employed and the degree of disturbance.

6.6.1 Effects on Geomorphology and Sediment Transport

Like sand and gravel mining, mineral mining can have a significant effect on channel morphology, depending upon the extraction method. General effects of mining, including increased sedimentation, accelerated erosion, change in substrate, and increased streambed and streambank instability have been discussed in Section 6.5.1. Mineral mining can have some additional effects on channel formation and stability. During dredging operations, gravels are removed from rivers but are not hauled away from the channel; gold is extracted and waste gravels are piled along the banks, covering the riparian vegetation. These piles may eventually revegetate but remain unstable and leave banks with a high potential for erosion (Nelson et al. 1991). Dredging for gold in the early 1900s left extensive mine tailings, which continue to constrict stream channels and serve as chronic sediment sources (McIntosh et al. 1994b). Records from the 1940s indicate that substantial portions of the upper Grande Ronde river flowed beneath the extensive rubble left behind be early mining operations (McIntosh et al. 1994b). Hydraulic mining, which involves washing of unconsolidated ore-bearing alluvial gravels out of river banks or from hillslope areas down into the river, is uncommon today, yet effects are still being propagated throughout many river systems from long-abandoned operations (AFS 1988). Several forms of mining (strip, open-pit, quarry) remove the vegetation and topsoils from the site creating the potential for erosion and increased sedimentation. If topsoils are not retained to cover mine spoils, revegetation may be inhibited for extended time periods, especially if mine spoils are acidic (Butterfield and Tueller 1980; Fisher and Deutsch 1983).

6.6.2 Effects on Hydrology

Mineral mining may alter the timing and routing of surface and subsurface flows. Surface mining may increase streamflow and storm runoff (Sullivan 1967; Collier et al. 1970), as a result of compaction of mine spoils, reduction of vegetated cover, and the loss of organic topsoils, all of which reduce infiltration (Nelson et al. 1991). Merz and Finn (1951 in Nelson et al. 1991) reported infiltration rates of 452.1 $cm \cdot h^{-1}$ on undisturbed soils versus 43.2 $cm \cdot h^{-1}$ on adjacent graded spoils banks. Lower infiltration rates mean overland runoff and streamflow increase, particularly during storm events. Increasing flows may cause channel adjustments, including increased width and depth. Pit and strip mining may also affect groundwater by physical disruption of aquifers (Nelson et al. 1991). Large amounts of water are needed for processing mining products, and in arid regions east of the Cascades, withdrawals for mining may significantly affect the limited water supplies. Lindskov and Kimball (1984) estimated that extraction of 400,000 barrels of oil annually from oil shales in Utah, Colorado and Wyoming would require 86 million m^3 of water per year, which would be pumped from groundwater aquifers (Nelson et al. 1991). It was the tremendous demand for water by mining operations that stimulated the water law of prior appropriation in the West (Wilkinson 1992).

6.6.3 Effects on Thermal Energy Transfer and Stream Temperature

Dredging and other mining practices may cause loss of riparian vegetation and changes in heat exchange, leading to higher summer temperatures and lower winter stream temperatures. Bank instability can also lead to altered width-to-depth ratios, which further influences temperature (see Section 3.7).

6.6.4 Effects on Nutrients and Pollutants

No published information was found regarding the effects of mineral mining on nutrient cycling or availability. However, surface mining and dredging likely affect inputs of nutrients where vegetation is removed or buried, and may increase nutrient spiraling length within streams where structure is simplified and nutrient retention is diminished.

Perhaps the most important effect of mining on aquatic ecosystems is contamination of surface waters from mine spoils. Acidification of surface waters by mining operations is generally considered to be the most serious consequence of mining. Water is acidified by oxidation of iron-containing waste products, which are then carried with runoff into local drainages (Nelson et al. 1991). In the western USA, much of the mineral recovery occurs from granitic deposits containing pyrite (Nelson et al. 1991). When exposed to atmospheric oxygen, pyrite is readily oxidized in water to produce sulfuric acid, which lowers the pH of mine spoils. Other metallic sulfides including chalcopyrite ($CuFeS_2$), sphalerite (ZnS), galena (PbS), and greenockite (CdS), undergo similar acid-generating processes (Nelson et al. 1991). Reductions in pH increase the mobility of many heavy metals (e.g., aluminum, arsenic, cadmium, chromium, cobalt, copper, iron, lead, mercury, nickel, zinc) by altering their chemical form, particularly if drainage is through waste piles. High acidity also facilitates formation of ferric hydroxide ($FeOH_3$), a noxious precipitate often called "yellow boy" (Nelson et al. 1991). The process of acidification is ongoing, and increased soil acidity converts metals into forms that are more bioavailable.

Nelson et al. (1991) reviewed the literature and found several examples of pollution associated with mine wastes. Levels of copper and zinc were 4–10 times above background levels 560 km from the major source of contamination on the Clark Fork River, Montana (Johns and Moore 1985). Duamie et al. (1985) reported that loadings from an abandoned mine in Montana were 13.6 $kg \cdot d^{-1}$ for copper and 1.6–145.5 $kg \cdot d^{-1}$ for zinc. Acid mine drainage and copper loadings of 41–147 $kg \cdot d^{-1}$ have been documented for Panther Creek, Idaho (Reiser 1986).

Heap leach mining is a form of open-pit mining used to extract gold from low-grade ore deposits. Extracted ore is crushed and placed into piles called pads where a dilute solution of sodium-cyanide ($NaCN$) is sprayed over the ore. As the cyanide solution percolates through the pad, gold is bonded to solutes and is collected in catch basins. With further processing, termed flotation, the gold is recovered. Cyanide is a well known toxicant, and any that leaches into local streams or is released from storage lagoons is potentially lethal to all aquatic organisms.

6.6.5 Effects on Physical Habitat Structure

The effects of surface mineral mining on the physical structure of salmonid habitats are similar to the effects of gravel and sand mining. Elevated levels of erosion increase sedimentation, which in turn affects the structure and composition of instream substrate. Spaulding and Ogden (1968) estimated that hydraulic mining for gold in the Boise River basin, Idaho produced 116,500 tonnes of silt in 18 months. They also reported that dredging in the Salmon River produced enough silt to cover 20.9 km of stream bottom with 0.16 cm of silt every 10 days, which reduced salmon spawning by 25%. Other effects of increased sedimentation include shallowing and widening of channels and reduction in pool frequency.

Dredging and placer mining practices have significantly altered the stability of habitats for fish and other organisms. An unnatural forced meander pattern was created along some sections of the Crooked River in Idaho, while another section was straightened. All along these disturbed sections, meadows and riparian vegetation were lost as a result of gold dredging (Nelson et al. 1991).

6.6.6 Effects on Stream Biota

Aquatic communities are affected by mining activities primarily through the alteration of physical processes (e.g., increased sediment inputs, greater channel instability, and simplification of channel structure) and chemical characteristics (e.g., acidification, heavy metals). Toxic effects of metals and acid can affect growth, reproduction, behavior, and migration of salmonids, resulting in the loss of sensitive species, changes in productivity, and alterations in population structure (AFS 1988). Increased turbidity reduces light penetration and decreases production of benthic algae (Nelson et al. 1991). Acidification of surface waters precipitates ferrous hydroxide, further decreasing benthic algal production and degrading macroinvertebrate habitat.

Stream acidification affects organisms that are sensitive to low pH, including salmonids. Salmonids exposed to low pH have been shown to experience reduced egg viability, fry survival, growth rate, development of pigmentation, ossification, and heart rate (Trojnar 1977; Nelson 1982). Johnson and Webster (1977) reported that spawning brook trout avoid areas of low pH, and speculated that recruitment is likely affected. Reduced numbers and diversity of benthic invertebrate taxa were found below an abandoned gold and silver mine on Coal Creek, Colorado (Reiser et al. 1982). Ephemeropterans (mayflies), plecopterans (stoneflies), and trichopterans (caddisflies) were found most sensitive to lowered pH in a study by Roback and Richardson (1969).

Lowered pH also enhances the availability and toxicity of heavy metals or metaloids. Arsenic, cadmium, chromium, cobalt, copper, iron, lead, manganese, mercury, nickel, and zinc, are all toxic to fish, affecting growth, metabolism, respiration, reproduction, and numerous other biological functions (reviewed in Nelson et al. 1991). These substances may act singly, in combination, synergistically, or antagonistically (to reduce toxicity). Since many of these metals tend to bioaccumulate, increased toxicity is seen in higher-level trophic organisms for a given "background level" in surface waters. Numerous studies have developed LD„ levels for these toxicants using various invertebrate and fish (Table 6-7) test organisms. Other studies have also shown that continuous exposure to sublethal levels may produce effects that are just as important for determining ultimate species survival in the affected habitat (EPA 1986).

Effects of chronic pollution from mine wastes have been documented for several streams in the west. Mining wastes containing arsenic, cadmium, copper and zinc have been contaminating the Clark Fork River in Montana for more than 125 years. These metals have resulted in elevated metal concentrations in stream biota (Woodward et al. 1993) and are believed to be affecting benthic invertebrate communities and trout productivity in the river (Pascoe et al. 1993). Laboratory experiments in which rainbow trout fry were exposed to metal concentrations in water and food comparable to those in the Clark Fork indicate that uptake through the diet was the more important source of exposure (Woodward et al. 1993). Exposed fish experienced reduced growth and survival compared to control fish. Hughes (1985) found that Montana streams subjected *to* periodic mine effluents had fundamentally altered benthos assemblages and reductions or elimination of trout and sculpins. Other examples of exposure of salmonids and other aquatic organisms to pollution from mine wastes are reviewed in (Nelson et al. 1991).

6.7 Effects of Hydroelectric Dams

Hydroelectric dams have contributed substantially to the decline of salmonids in the Pacific Northwest, particularly anadromous stocks in the Columbia, Snake, and Sacramento River systems. The Northwest Power Planning Council (NPPC) estimates that current annual salmon and steelhead production in the Columbia River Basin is more than 10 million fish below historical levels, with **8** million of this annual loss attributable to hydropower development and operation (NPPC 1986). They conclude that approximately half of these losses occur during fish passage through the mainstem projects below Chief Joseph Dam (upper Columbia River) and Hells Canyon Dam (Snake River). The remaining 4 million in losses are due to the restriction of the fishes' range caused by dams; access to approximately 55% of the total basin area and 33% of the linear stream miles has been blocked by dams (Thompson 1976; PFMC 1979).

Dams influence salmonids and their habitats in a variety of ways. They impede migration of juvenile and adult fish, delaying migration (Raymond 1979) and thereby increasing the duration of exposure to predators. Juvenile or adult fish that pass through turbines may be killed outright or may be injured or disoriented, becoming easy prey for aquatic and terrestrial predators. Attempts to bypass dams by barging or trucking may stress fish and increase disease transmission among individuals (Bevan et al. 1994a), which ultimately may reduce survival.

Hydroelectric operations alter natural flow regimes, including daily and seasonal flow patterns. Unnaturally large daily fluctuations in flow occur downstream of dams during peaking operations. Seasonal flow fluctuations tend to be dampened, with water stored during periods of high flow in the winter or spring and released in summer when natural flows are lower (Marcus et al. 1990). These changes can affect migratory behavior of juvenile salmonids. Water-level fluctuations associated with hydropower peaking operations may reduce habitat availability, inhibit the establishment of aquatic macrophytes that provide cover for fish, and in some cases strand fish or allow desiccation of spawning redds (Palmisano et al. 1993a). The impoundment of water behind dams creates slackwater environments that are less favorable to salmonids. With the exception of the Hanford reach, virtually the entire lower and mid-Columbia River has been changed from a free-flowing river to a series of ponded reaches with little fast water, and significant spawning areas have been lost. The slow-moving water behind impoundments are also favorable to certain predators of salmonids, including northern squawfish (Faler et al. 1988).

Table 6-7. Reported toxicities of metals in soft water (< 45,000 $\mu g \cdot L^{-1}$ as $CaCO_3$). From Nelson et al. (1991). Reproduced with permission from the publisher.

Substance	Species	Reported Toxicity		Source
		Method*	Concentration ($\mu g \cdot L^{-1}$)	
Aluminum (Al)	Brook trout	LC50	3,600 – 4,000	Decker and Menendez (1974)
Arsenic (As)	Rainbow trout	LC50	10,800	Hale (1977)
Cadmium (Cd)	Rainbow trout	LC50	6.6	Hale (1977)
	Brook trout	MATC	1.7 – 3.4	Benoit et al. (1976)
Chromium (Cr)	Rainbow trout	LC50	24,100	Hale (1977)
	Rainbow trout	LC50	69,000	Benoit (1976)
	Rainbow trout	MATC	200 – 350	Benoit (1976)
	Brook trout	LC50	59,000	Benoit (1976)
	Brook trout	MATC	200 – 350	Benoit (1976)
Copper (Cu)	Coho salmon	LC50	46.0	Chapman and Stevens (1978)
	Rainbow trout	LC50	253.0	Hale (1977)
	Rainbow trout	LC50	125.0	Wilson (1972)
	Rainbow trout	LC50	57.0	Chapman and Stevens (1978)
	Rainbow trout	ILL	37.0	Sprague and Ramsay (1965)
	Atlantic salmon	ILL	32.0	Sprague and Ramsay (1965)
	Atlantic salmon	ILL	520	Sprague (1964)
	Brook trout	MATC	9.5 – 17.4	McKim and Benoit (1971)
Copper-zinc (Cu-Zn)	Atlantic salmon	TU	1,000	Sprague and Ramsay (1965)
Iron (Fe)	Brook trout	LC50	1,750	Decker and Menendez (1974)
Lead (Pb)	Rainbow trout	LC50	8,000	Hale (1977)
	Rainbow trout (eggs)	MATC	4.1 – 7.6	Davies et al. (1976)
Mercury (Hg)	Rainbow trout	LC50	33.000	Hale (1977)
Nickel (Ni)	Rainbow trout	LC50	35,500	Hale (1977)
Uranium (U)	Brook trout	LC50	2,800	Parkhurst et al. (1984)
Zinc (Zn)	Coho salmon	LC50	905.0	Chapman and Stevens (1978)
	Rainbow trout	LC50	1,775	Chapman and Stevens (1978)
	Rainbow trout	LC50	180 – 390	Finlayson and Ashuckian (1979)
	Rainbow trout	ILL	560	Sprague and Ramsay (1965)
	Atlantic salmon	ILL	92	Sprague (1964)
	Atlantic salmon	ILL	150 – 1,000	Zitko and Carson (1977)
	Atlantic salmon	ILL	420	Sprague and Ramsay (1965)
	Brook trout	MATC	534 – 1,360	Holcombe et al. (1979)

*LC50 = lethal concentration for 50% of test organisms; MATC = maximum acceptable toxic concentration; ILL = incipient lethal level: TU = toxic units.

Hydroelectric dams also modify sediment transport, natural temperature regimes, and the concentration of dissolved gases. Water storage at dams may prevent flushing flows that are needed to scour fine sediments from spawning substrate and move wood and other materials downstream. Behind dams, suspended sediments settle to the bottoms of reservoirs, covering coarser substrate **and** depriving downstream reaches of needed sediment inputs. The reduction in sediments downstream of dams leads to changes in channel morphology (Marcus et al. 1990). Reservoirs also modify temperature regimes in streams and rivers. Below larger reservoirs that thermally stratify and that have hypolimnetic discharges, seasonal temperature fluctuations generally decrease; temperatures are cooler in the summer as cold hypolimnetic waters are discharged, but warmer in the fall as energy stored in the epilimnion during the summer is released. Finally, dams have resulted in changes in concentrations of dissolved oxygen and nitrogen concentrations (Bevan et al. 1994a). Behind dams, slow-moving water has lower dissolved oxygen levels than faster, turbulent waters. Water that spills over dams entrains air, and supersaturation of dissolved gases results. Gas supersaturation can cause gas bubble disease in salmonids, resulting in mortality, or weakening fish such that they become more vulnerable to predation or infection (Parametrix 1975; Blahm et al. 1975).

An exhaustive review of effects of dams on salmonids is beyond the scope of this document. A more thorough discussion of effects of dams on endangered salmonids in the Columbia Basin can be found in the recovery plan for Snake River salmon (Bevan et al. 1994a).

6.8 Effects of Irrigation Impoundments and Withdrawals

Damming and diversion of streams and rivers for agricultural purposes began in earnest in the mid-1800s as settlers moved into the region (Wilkinson 1992; Palmisano et al. 1993a). In the Pacific Northwest, withdrawals for agriculture (crop irrigation and stock watering) currently account for the vast majority (80%–100%)of offstream water uses in all major sub-basins east of the Cascades and in the upper Klamath Basin (Muckleston 1993). In addition, agriculture accounts for 62% of offstream water use in the coastal basins of Oregon, and 28% of the use in the Willamette Valley.

Water for irrigation is withdrawn in several ways. For major irrigation withdrawals, water is either stored in impoundments or diverted directly from the river channel at pumping facilities. Individual irrigators commonly construct smaller "push-up" dams from soil and rock within the stream channel, to divert water into irrigation ditches or to create small storage ponds from which water is pumped. In addition, pumps may be submerged directly into rivers and streams to withdraw water.

Many of the effects of irrigation withdrawals on aquatic systems are similar to those associated with hydroelectric power production, including impediments to migration, changes in sediment transport and storage, altered flow and temperature regimes, and water level fluctuations. In addition, aquatic organisms may be affected by pollutants from agricultural runoff and reduced assimilative capacity of streams and rivers from which substantial volumes of water are withdrawn. Alterations in physical and chemical attributes in turn affect many biological components of aquatic systems including vegetation within streams and along reservoir margins, as well as the composition, abundance, and distribution of macroinvertebrates and fishes.

6.8.1 Fish Passage

For many early irrigation dams, no fish passage facilities were constructed, resulting in the loss of several significant salmon runs. For example, irrigation dams in the Yakima River basin blocked sockeye runs estimated at 200,000 adult fish (Palmisano et al. 1993a). At some older irrigation impoundments (e.g., the Savage Rapids Dam on the Rogue River in Oregon), adult passage is hindered by poorly designed fish ladders (BR 1995). Smaller instream diversions may also impede the migrations of adult fish or cause juveniles to be diverted into irrigation ditches. Salmonid juveniles and smolts are also lost through entrainment at unscreened diversions or impingement on poorly designed screens.

6.8.2 Flow Modifications and Water-Level Fluctuations

The volume of water diverted for agriculture is substantial. Muckleston (1993) reports that withdrawals in the Snake River basin total approximately 45,000 acre-feet per day (equivalent to approximately 636.8 $m^3 \cdot s^{-1}$ (22,500 cfs); because this value is an annual average, daily diversions during the peak irrigation season are likely much higher. Diversion from individual rivers may also be great. For example, the Wapato Canal has a capacity to withdraw 57 $m^3 \cdot s^{-1}$ (2,000 cfs) from the Yakima River, with operation usually extending from March to mid-October (Neitzel et al. 1990).

Irrigation withdrawals affect both the total volume of water available to fish and the seasonal distribution of flow. Dams for irrigation typically store water during periods of high runoff in the winter or spring, and release water during the summer when flows are naturally low. Consequently, these impoundments tend to moderate streamflows, reducing winter and spring peak flows. Most direct diversions from rivers occur from spring to fall, during the peak growing season of agricultural crops. Because irrigation of crops coincides with periods of maximum solar radiation, evapotranspiration losses are greater than would occur under normal rainfall-runoff regimes, resulting in reduced summer flows in streams and rivers.

Changes in the quantity and timing of streamflow alters the velocity of streams which, in turn, affects all types of aquatic biota. Water velocity is a major factor controlling the distribution of periphyton and benthic invertebrates in streams (Hynes 1970; Gore 1978; Horner 1978). At low velocities, diatom-dominated periphyton communities may be replaced by filamentous green algae (McIntire 1966). In western Washington streams, periphyton growth rates increased as velocity increased up to 0.1 $m \cdot s^{-1}$ (Gore 1978); however, as velocities increase above that level, erosion of periphyton exceeds growth. Reduced velocity may eliminate invertebrate species that require high velocities (Trotzky and Gregory 1974). The abundance and composition of fish species and assemblages is also regulated by the water velocity (Powell 1958; Fraser 1972). Changes in velocity influence incubation and development of eggs and larval fish by affecting oxygen concentrations within the gravel (Silver et al. 1963). Reduced water velocities in the Columbia River, which are in part a result of agricultural diversions, may delay downstream migration of salmon smolts. If temperatures become excessively warm, smolts may discontinue migration and revert to a presmolt physiology (Ebel 1977). Survival of these holdovers (fish delaying seaward migration for a year or more) is only about 20% (Adams et al. 1975), and very few may survive to return as adults (CRFC 1979).

Where irrigation water is withdrawn from smaller streams, seasonal or daily flow fluctuations may affect fish, macroinvertebrates in littoral areas, aquatic macrophytes, and periphyton (reviewed in Ploskey 1983). Lowered water levels may concentrate fish, which potentially increases predation and competition for food and space (Aggus 1979). Fluctuating water levels may delay spawning migrations, impact breeding condition, reduce salmon spawning area (Beiningen 1976), dewater redds and expose developing embryos, strand fry (CRFC 1979), and delay downstream migration of smolts. Water level fluctuations in reservoirs also reduce the density of bottom-dwelling organisms (Fillion 1967; Stober et al. 1976; Kaster and Jacobi 1978) through stranding, desiccation, or exposure to freezing temperatures (Powell 1958; Kroger 1973; Brusven and Prather 1974). In the littoral zone, frequent changes in water level can eliminate aquatic macrophytes that provide habitat for fish (Munro and Larkin 1950; Aas 1960). Loss of periphyton (attached algae) in the stream margins because of desiccation has been observed below hydroelectric dams (Neel 1966; Radford and Hartland-Rowe 1971; Kroger 1973) and may occur along the margins of streams below pumping facilities. Reductions in periphyton production affects other levels in the food web, particularly in large, unshaded rivers, where periphyton can be an important energy source.

6.8.3 Changes in Sediment Transport

Irrigation withdrawals and impoundments can affect the quantity of sediments delivered to streams and transported down river. In general, siltation and turbidity in streams both increase as a result of increased irrigation withdrawals because of high sediment loads in return waters. Unlined return canals contribute heavier silt loads than lined canals or subsurface drains (Sylvester and Seabloom 1962). Turbidity in the Wenatchee River doubled over a 45-year period because of increased agriculture and other human activities (Sylvester and Ruggles 1957). Once in the stream channel, the fate of sediments depends on hydrologic conditions. In systems where total water yield or peak discharge are reduced, sediments may accumulate in downstream reaches, affecting the quality of salmonid habitats. In the Trinity River in California, extreme streamflow depletion (85%–90% of average surface runoff) has allowed sediments to accumulate downstream, covering spawning gravels and filling in pools that chinook salmon use for rearing (Nelson et al. 1987). The lack of flushing flows during the winter has exacerbated this problem. In other systems, concentrations of suspended sediments below irrigation impoundments may be lower because slower water velocities allow sediments to settle (Sylvester and Ruggles 1957). The deposition of coarse, gravel sediments may be essential for developing high quality spawning gravels downstream of impoundments. Downstream reaches may become sediment starved, and substrate is frequently dominated by cobble and other large fractions unsuitable for spawning.

Iwamoto et al. (1978) reported that algae, phytoplankton, zooplankton, benthos, aquatic insects, and fish are all adversely affected by suspended and shifting sediments. In addition, sediments deposited into reservoirs, coupled with reduced streamflows, may improve habitat for intermediate hosts of several fish parasites. The impacts of suspended sediments, turbidity, and siltation are discussed in greater detail in Section 5.1.2.

6.8.4 Changes in Stream Temperature

Irrigation impoundments and withdrawals may increase water temperatures by increasing the surface area of rivers (i.e., reservoirs), reducing discharge volume, and returning heated irrigation waters to streams. In systems with irrigation impoundments, the seasonal thermal regime may also be altered. Reservoirs allow heating of surface waters that, depending on whether releases are from the epilimnion or hypolimnion, can result in increased or decreased temperatures. Below Lost Creek Dam on the Rogue River in Oregon—a multipurpose dam from which irrigation waters are withdrawn—

temperatures decreased during summer because of hypolimnetic discharges but increased during the autumn and winter as water that had been heated during the summer was released (Satterthwaite et al. 1992). The increases in fall and winter temperatures accelerated embryonic development of chinook salmon, resulting in earlier emergence. Typically, return flows of surface water from irrigation projects are substantially warmer after passage through the canals and laterals common to irrigated agriculture (Sylvester and Seabloom 1962). The degree to which water temperatures are affected by withdrawal of irrigation water ultimately depends on the proportion of water removed from and returned to the system and on the seasonal hydrologic regime. Water withdrawals in years of low flow are likely to have greater thermal effects on the fishes and other aquatic biota compared with similar withdrawals during years of high flow.

6.8.5 Changes in Dissolved Oxygen

Dissolved oxygen (DO) concentrations may decrease in both summer and winter in systems with irrigation withdrawals or impoundments. During summer, high solar radiation and warm air and ground temperatures combine to raise the water temperature of irrigation return flow, which diminishes the ability of water to hold DO. Increased water temperatures of irrigation return flows have been shown to reduce DO levels in the Yakima River (Sylvester and Seabloom 1962). Low summer flows can allow greater diel temperature fluctuations, which may exacerbate reductions in DO (McNeil 1968). In addition, higher concentrations of nutrients associated with irrigation returns may reduce DO by increasing biochemical oxygen demand. The extent and period of reduced DO concentrations depends on the quantity of water withdrawn and the quality of the return flow. In winter, low DO levels may occur in irrigation impoundments that have been drawn down. Fish kills can occur through anoxia if lowered water level facilitates freezing, which in turn inhibits light penetration and photosynthesis (Ploskey 1983; Guenther and Hubert 1993).

6.8.6 Influence of Impoundment and Water Withdrawal on Fish Diseases

Impoundment and water withdrawal for off-stream use may facilitate disease epizootics in salmonids by altering temperature regimes, lowering water levels, reducing flow velocities, creating habitat for intermediate hosts of parasites, and concentrating organisms, thereby facilitating the transmission of certain pathogens. Pathogen virulence and salmonid immune systems are greatly affected by water temperature (see Section 4.3.4); thus increasing temperatures by impoundment, flow reduction, or return of heated irrigation waters will affect disease susceptibility and prevalence in fish populations. Becker and Fujihara (1978) emphasize that extended periods of warm temperature and low flow increase the epizootiology of *F. columnaris* in Columbia River fish populations, and they warn that increasing withdrawal of Columbia River water for offstream use increases the potential for disease. Bell (1986) suggests that fish populations inhabiting lakes and reservoirs tend to experience more disease epizootics than fish species found in free-flowing rivers. Diseases in impoundments generally occur as a result of widespread parasite infections (Bell 1986). Decreasing water depth may provide additional habitat for intermediate hosts of parasites. Snail populations, as well as parasitic trematodes that use snails as intermediate hosts (e.g., *Diplostomum* and *Posthodiplostomum*), are more abundant in shallow waters (Hoffman and Bauer 1971). Consequently, reductions in flow may increase the likelihood of parasite epidemics. Finally, return flows from irrigated fields may transport parasitic nematodes and viruses from infested fields into streams (BR 1976).

6.9 River, Estuary, and Ocean Traffic (Commercial and Recreational)

Within a few decades of settlement, many estuaries and large low gradient rivers of western California, Oregon, and Washington were channelized. Eventually significant portions of major rivers, including the Columbia and Sacramento, were radically transformed.These systems were first altered for riverboat navigation and later to accommodate log rafting, barges, and ports (Maser and Sedell 1994). Navigation channels and pools continue to be maintained by dredging, removing snags, installing revetments, and operating locks and dams. Consequently, salmonids evolved in rivers and estuaries much different from what we now see in most of the Pacific Northwest.

What once was an incredible complex of channels, islands, bays, and wetlands connected with the sea are now highly simplified conduits. These complex mazes of shifting channels and bars laden with enormous snags and jams impeded navigation, but they were a haven for resident and migrating salmonids. Braided channels under gallery forests and flowing through alluvial plains with high water tables had abundant inflows of cool ground water during the summer. Water was stored in extensive floodplains (instead of behind dams) during the wet season and entered the channel via subsurface flows during the dry season. Network of essentially small, partially shaded rivers offered much more productive rearing habitat than the present navigation channels. These complex channels were nutrient-rich rearing habitats in the summer and provided refugia from high streamflows during the winter. Changes in channel complexity are exemplified by the 80% reduction in the number of upper mainstem Willamette River channels documented by Sedell and Froggatt (1984).

High flows that once signaled migrations, offered passage over falls, and transported smolts rapidly to the estuaries have been moderated to facilitate the year-round boat traffic. For example, a 1938 Oregon statute directs that a year-round minimum of 6,000 cfs be maintained in the Willamette River at Salem (Muckleston 1993). As a consequence, numerous flood control reservoirs are operated to ensure this flow. The harbors, docks, and marinas offer some limited cover, but salmonids may be exposed to high levels of wood preservatives, petroleum, and organic wastes, as well as intense angling pressure. Noise pollution from boat traffic in estuaries and the open ocean may disrupt the navigation and communication of sharks and toothed whales; this may indirectly affect salmonids because sharks and whales feed on seals and sea lions, which in turn prey on salmon.

Channel changes have markedly altered the abundance and distribution of salmonids by making the physical habitat less suitable to resident and migrating fish. Losses of these salmon produced losses of particular life-history strategies from the population. Moreover, because these low gradient habitats were also among the most productive freshwater areas for salmon, their degradation has perhaps resulted in the loss of more fish than alterations of higher gradient reaches of similar size.

6.10 Wetland Loss/Removal

In 1989, Congress directed the Secretary of the Interior to assess the estimated total acreage of wetlands in each State in the 1780s and in the 1980s. The study (Dahl 1990) estimated that approximately 89.44 million hectares (221 million acres) of wetland functioned in the conterminous United States in the 1780s and that 53% of that area had been lost by the 1980s. Wetlands lost during this period included 1,839,741 hectares (4,546,000 acres) in California, 198,826 hectares (491,300 acres) in Idaho, 351,315 hectares (868,100 acres) in Oregon, and 166,734 hectares (412,000 acres) in Washington. These losses meant wetland area decreased from 4.9% to 0.4% of the land area in California, 1.6% to 0.7% of the land area in Idaho, 3.6% to 2.2% of the land area in Oregon, and 3.1% to 2.1% of the land area in Washington. These losses changed the function of ecosystems at the landscape scale because wetlands affect the transport and character of water in watersheds, lakes, and streams.

Wetlands provide a moderated climate compared to the adjacent uplands (cooler in summer and warmer in winter) because of the ground water (at relatively constant temperature) supplied to the site and the microclimate that develops within the vegetation occupying the wetland. Activities that modify the ground water supplied to the site, or modify the plant community, can impair the wetland's ability to moderate climate. Wetlands typically occur as a transition between upland and aquatic ecosystems, for example, at the edge of streams or between the stream and the adjacent valley walls. Wetlands require the surplus water that distinguishes them from uplands (EPA 1980b). Because wetlands may be only slightly wetter than adjacent upland, they are often targeted for drainage—either by ditching or tiling. These activities change the timing and duration of wetness of the site and modify or impair the wetland's functions. Diking may cause wetlands to be drier where the dike prevents floodwater from entering the wetland. Diking also may eliminate some functions performed by the wetland, for instance, floodwater storage; however, most wetlands do not exclusively depend on floodwater for their existence. Consequently, diking may not totally eliminate other normal wetland functions (as described below). Building, paving, or other permanent changes to the wetland's surface usually eliminate the majority of its functions, although some functions (e.g., floodwater storage) may continue at the site. Wetlands perform several functions related to hydrology, water quality, and habitat; these functions ultimately support salmonids.

6.10.1 Wetlands and Hydrology

Wetlands store water during runoff events, thereby reducing flood volumes and flood stages downstream. Further, floodwaters slow as they move into wetlands, reducing damage associated with scour and erosion caused by high velocity flows and allowing sediments, particulate organic matter, and other materials to be deposited in the wetland. Water quality improves with deposition of sediments, and some dissolved materials are either trapped within sediment deposits or utilized by vegetation and organisms in the wetlands. Movement of water through the wetland may also redistribute organic and inorganic particulates as well as import or export plant propagules or organisms. Infiltration of the flood waters into wetland soils supports other wetland functions, such as nutrient cycling, the retention and processing of elements and compounds, and the support of microbial communities adapted to survival in anaerobic conditions. And finally, because of unique hydrologic characteristics and soils, wetlands support unique floral and faunal communities. Wetlands are an integral component in the hydrologic cycle locally and of the habitat provided by the total watershed.

Novitzki (1979) reported that wetlands had a pronounced influence on flood peaks and seasonal distribution of streamflow. In Wisconsin, flood flows were 80% lower in watersheds with 40% lake-and-wetland area than in watersheds with no lake-or-wetland area. (Wetlands occupied 14.8% of the land area of Wisconsin in the 1980s [Dahl 1990], so wetlands may have a greater influence on streamflow in Wisconsin than in the Pacific Northwest.)

Wetlands also modify the rate of ground-water discharge to streams (Novitzki et al. 1993). Wetlands, particularly those occurring adjacent to streams, usually exist because of ground-water discharge. Wetland soils typically are less permeable than upland soils, especially where prolonged wetness fosters the accumulation of organic material (Novitzki 1989). Because the wetland soils are less permeable, the rate of ground-water discharge from upgradient sources, through the wetland, and to the stream is slowed. The net effect is to reduce the rate of ground-water discharge to the stream but to increase the length of time that discharge occurs. Ground water typically discharges at a relatively constant, cool temperature, and it has a major influence on the temperature regime in streams, especially low-flow periods in summer. Changing the amount and timing of ground-water discharge may change the temperature regime of the stream significantly, affecting the suitability of the stream as salmon habitat. Ground-water upwelling into streams through gravels is a determinant in selection of spawning redd for some species. The constant upwelling of fresh, oxygenated water may be necessary for egg and fry survival. Loss of wetlands will likely change the rate of ground-water discharge at critical times and may reduce spawning success in streams.

6.10.2 Wetlands and Water Quality

Wetlands retain particulate materials transported into them by overland flow or river flooding. Wetlands typically are flat areas adjacent to streams, and as floodwaters enter, flow velocities decrease and sediment loads are deposited. This phenomenon manifests as berms, often wooded, that build up next to the river channel in wide river valleys. Wetlands tend to stabilize stream banks because of the robust plant community that grows there. Wetlands tend to be wet through a larger part of the growing season, fostering plant growth that in turn provides sufficient root mass to stabilize soils. Where banks are stabilized by the lush wetland vegetation, stream channels tend to be somewhat deeper, and undercutting provides shelter to salmonids and other aquatic biota. Logging, grazing, farming, or other activities that change the wetland plant community can significantly reduce the wetland's ability to stabilize stream banks. Moreover, the velocity of water moving through wetlands is further reduced by dense vegetation, especially shrubs and trees, which in turn increases sediment deposition in the wetland. Thus, wetlands tend to reduce the amount of sediment transported to streams. Loss or removal of wetland areas may result in increased sediment loads (especially clays and silts) in receiving streams.

Wetlands also retain and process dissolved materials contained in overland flow or floodwaters. Some nutrients, as well as toxic substances, are taken up by plants, while others are bound to suspended solids, which subsequently settle to the bottom. Thus, loss or removal of wetland areas may result in increased nutrient and contaminant loading to receiving streams.

6.10.3 Wetlands and Salmonid Habitat

Wetlands may contribute significantly to certain characteristics required by salmonids in their aquatic ecosystems, such as variable, but moderate streamflows; cool, well oxygenated, unpolluted water; relatively sediment-free streambed gravel; an adequate food supply; and instream structural diversity provided by woody debris (Cederholm 1994). Because wetlands affect flood flows and springtime flows, they also influence the streamflow characteristics of the streams and aquatic habitat that support salmonids. Loss of wetlands likely increases the amount of individual flood peaks but reduces the duration of high-flow events. Streams in the Pacific Northwest may require the infrequent (i.e., the 100-year) flood to reset; however, they may also require stability between extreme events to recover and re-establish equilibrium. Wetland loss may reduce the time between significant (e.g., 5- to 50-year frequency) floods and impair the stream's ability to recover. For some salmonids, the timing and amount of streamflow triggers the movement of adult salmon into spawning streams, as well as the movement of fry and smolts downstream. Changing the timing of lows may thus subtly change the timing of migration and spawning, resulting in disruption of natural biological cycles. Changing the timing of spawning may result in minor, but significant, changes in the size and condition of salmon smolts returning to the ocean. These changes may have pronounced impacts on survival of young salmon in the ocean phase of their life cycle.

Wetlands support unique floral and faunal communities. The unique biota supported in wetlands contribute to the food web supporting the salmonids and associated biota in the streams, both adjacent to and downstream of the wetlands. Riparian vegetation, including that in wetlands, regulates the exchange of nutrients and material from upland forests to streams and wetlands (Cederholm 1994). Wetlands and ponds have been found to provide critical habitats for both juvenile salmonids (Peterson 1982; Cederholm and Scarlett 1982) and a variety of wildlife species (Zarnowitz and Raedeke 1984). Species that frequent riparian areas include amphibians, reptiles, birds, mammals, and mollusks (FEMAT 1993). Activities that prevent the normal wetland functions or impair the connectivity of the wetland to the aquatic ecosystem may prevent the transport of materials into and out of the wetland, altering important elements of aquatic ecosystems. Interrupting or otherwise changing the connections between the wetland and the stream can impede the exchange of nutrients, organic detritus, insects, or other materials supporting the food web of the aquatic ecosystem.

Wetlands often provide refugia within the landscape. Especially in urban areas, agricultural areas, or other disturbed environments, wetlands are least suitable for conversion to other use: they often are left intact until all other lands have been converted. In highly modified landscapes, they may be the only natural areas left to provide needed refuge to birds, mammals, and other biota. Because they typically occur at points of ground-water discharge, and reduce the rate but prolong the duration of ground-water discharge, wetlands also provide survival areas to aquatic species sensitive to high or low temperatures during hot summer periods or cold winter periods. In addition, a wetland may offer the only wet habitat available during periods of prolonged drought or during fires to protect those biota able to seek refuge within it.

Wetlands function as an integral component of the local watershed. They tend to be highly productive areas, often serving as a source of organic detritus to adjacent water bodies. Wetlands also provide nursery areas for salmon and habitat for organisms that provide food to salmon and associated biota. The wetland contributes to the ecological balance within the watershed/ecosystem within which it occurs. Destroying, draining, or otherwise impairing the wetland's function alters the hydrologic, sediment, chemical, and biological balance in the watershed.

6.11 Salmonid Harvest

Although this document focuses on the effects of human activities on salmonid habitats, it is essential to recognize the effects fisheries have had on salmonid populations in the Pacific Northwest. The harvest of salmonids by humans constitutes a significant source of mortality for both anadromous and resident species. For thousands of years preceding settlement of the West by Euro-Americans, Native Americans depended on salmonids as an important source of food, and salmonids continue to be central to the culture and economy of many tribes. Since the mid-1800s, large number of salmonids have been taken in off-shore and in-river commercial and recreational fisheries. In the late 1800s and early 1900s, chinook salmon dominated commercial landings off Washington, Oregon, and California (Deimling and Liss 1994), as well as in-river fisheries in the lower reaches of the Columbia, Sacramento-San Joaquin, and other large river systems. These fish were targeted for their large size and high food quality. Subsequently, salmonid fisheries have become progressively more diverse with other anadromous salmonids, particularly coho and pink salmon and steelhead trout, accounting for growing fractions of the total catch.

The relative importance of different fisheries varies across the region. From central California to Cape Flattery, Washington, ocean commercial troll and recreational fisheries account for the highest catch of anadromous salmonids, although substantial in-river sport and tribal harvest occurs in some river systems. The Pacific Fishery Management Council (PFMC 1995) estimates that between 1971 and 1990, combined catch of coho salmon in the commercial troll and recreational fisheries off Washington (Puget Sound included), Oregon, and California averaged over 1.9 million fish annually. Catch of chinook in the region during the same period averaged 1.3 million fish, and average catch of pink salmon in odd-numbered years (primarily in Washington) was about 200,000 fish. The Oregon Department of Fish and Wildlife (ODFW 1982) estimates that from 1971–1975 the commercial troll fishery accounted for 67% of the total coho salmon harvest in the Oregon Production Index (OPI) area (Columbia River south to central California), with the ocean recreational fishery and Columbia River gill-net fisheries accounting for 23% and 8%, respectively. In-river recreational and tribal fisheries made up the remaining 2% of the catch. The allocation of chinook salmon among various fisheries in Oregon varies with region and life-history types. For north-migrating stocks, ODFW allocates approximately 50%–55% of total annual harvest to the ocean troll fishery and approximately 45% to the in-river recreational fishery. For south-migrating stocks, ocean troll fisheries account for 60%–67% of the total fish harvested, whereas in-river fishery targets are approximately 15%–37% of total harvest (ODFW 1991).

In the Puget Sound area (Strait of Juan de Fuca, Puget Sound, Hood Canal, Nooksack–Samish, Skagit, and Stillaguamish–Snohomish units) commercial fisheries, including Indian and non-Indian, gill-net, purse seine, and troll fisheries, dominate the catch of salmonids: gill-nets and purse seines accounted for greater than 91% of the commercial catch in 1989 and 1990 (Palmisano et al. 1993a). Total commercial harvest in the Puget sound area compares with the combined troll and recreational catch from coastal waters along Washington to California, with pink salmon and sockeye salmon being numerically dominant. From 1971–1990, combined annual catch of pink salmon (odd-years only) in treaty and nontreaty commercial fisheries averaged approximately 2.9 million fish, while catch of sockeye salmon averaged over 1.8 million. During this same period, catch of coho, chum, and chinook salmon averaged 972,000, 768,000, and 211,000 fish, respectively (PFMC 1993). From 1979–1990, sport harvest in the Puget Sound area averaged approximately 8.6% of the commercial catch (Palmisano et al. 1993a).

Estimating total harvest rates on specific stocks or species of anadromous salmonids is difficult. These calculations require accurate estimates of 1) total ocean and in-river harvest (including harvest of fish originating in Oregon, Washington, and California by fisheries in Alaska and British Columbia), 2) spawning escapement (sometimes direct counts, but often estimated from index streams), 3) indirect hooking and dropout mortality, and 4) rates of natural mortality for species with a variable period of ocean residence. Despite uncertainty associated with each of these estimates, calculations of total harvest rates for several anadromous salmonid populations provide some indication of the magnitude of fishing effects. Between 1960 and 1983, harvest rates of coho salmon in the OPI area ranged from 57% to 87%, with a mean exploitation rate of 71%. Harvest rates were lower from 1984–1993, ranging from 27%–62% with a mean of 43% (T. Nickelson, ODFW, personal communication). Further reductions in harvest rates occurred in 1994 and 1995 because of the closure of the coho fisheries. Ocean harvest rates of chinook salmon originating from the Sacramento-San Joaquin system in California ranged from 50%–79% between 1970 and 1992, with a mean harvest rate of 67% (PFMC 1993); in-river fisheries were not included in these estimates. In its management plan for coastal chinook salmon stocks, the Oregon Department of Fish and Wildlife (ODFW 1991) concludes that most chinook populations can sustain harvest rates below 67% without compromising long-term conservation goals. Actual target harvest rates for various stocks of chinook and coho salmon are adjusted depending on stock strength and specific escapement goals. Recent evidence indicates that biased selection of index streams has resulted in overestimation of spawning escapement and, hence, underestimation of harvest rates (Cooney and Jacobs 1994). In addition, harvest targets consider only numerical abundance and do not address other long-term effects discussed below.

Adverse effects of harvest on salmonids are particularly difficult to control in mixed-stock fisheries, where multiple species, stocks, and age classes are harvested together. Mixed-stock fisheries occur primarily in the ocean and lower river reaches, before stocks segregate into discrete spawning runs. Mixed-stock fisheries are difficult to manage because escapement goals and harvest rates of different stocks constituting the fishery cannot be controlled. Consequently, strong and weak stocks are harvested at comparable rates, as are fish of wild and hatchery origin. For example, in the Columbia River system, where 90%–95% of the coho salmon are of hatchery origin, harvest rates approaching 90% still allow adequate escapement for hatchery brood-stock purposes, whereas ODFW (1982) estimates that harvest rates should be less than 69% to meet escapement goals for wild coho. Thus, where hatchery and wild coho salmon commingle in the ocean, wild fish are likely to be harvested at an excessive rate. Mixed-stock fisheries are especially detrimental to naturally small populations or populations that have been depressed by human activities. In these populations, escapement may be insufficient to maintain genetic diversity, and the probability of undesirable founder effects increases.

In addition to reducing total escapement of adult salmonids, harvest alters the age- and size-structure of salmonid populations. For example, Ricker (1981) provided evidence that mean sizes of all five Pacific salmon species harvested in British Columbia have decreased over the past 30 to 60 years. Similarly, between 1935 and 1989, the average weight of coho salmon caught in commercial fisheries off the coast of Washington declined by almost 30%; over the same period, mean weight of chinook, pink, and sockeye salmon decreased by 24%, 19%, and 14%, respectively (Palmisano et al. 1993a). Changes in size and age-structure arise for several reasons. For long-lived species that spend several years at sea, such as chinook salmon and steelhead trout, decreases in average size and age occur because immature individuals are harvested by troll fisheries over a number of years. Thus, larger and older individuals are harvested at a higher rate than individuals that mature earlier and at smaller size (Moussalli and Hilborn 1986), particularly for those stocks frequenting coastal waters rather than only passing through coastal waters on their way to spawning areas. Changes in size structure may also result from size-selective fishing gear. Ricker (1981) attributed decreases in average size of coho and pink salmon adults to cumulative genetic effects caused by selective removal of larger individuals in troll and gill-net fisheries. Selective removal of larger fish may also increase the percentage of "jacks" in sockeye salmon populations. In freshwater fisheries, size limits and gear restrictions also alter size and age-structure of salmonid populations. For example, Gresswell and Varley (1988) reported that mean length of cutthroat trout caught in Yellowstone Lake rose from a low of 365 mm in 1966–1967 to a high of 395 mm in 1983–1984 following a change in fishing restrictions from three fish of any size to two fish with a 330 mm maximum. Average age of spawners in Clear Creek, a major spawning tributary of the lake, increased from 4 years to almost 5.5 years during this period, with a concomitant increase in the frequency of fish aged 7–9.

Changes in average size and age of individuals influences success of salmonid populations in several ways. Large size provides salmonids with the energetic reserves needed to undertake extensive migrations as well as the ability to negotiate large barriers that are impassable to smaller fish; thus, the elimination of large individuals through harvest can effectively diminish the ability of populations to use particular spawning habitats. In addition, because the fecundity of salmonids typically increases with size, the selective removal of larger fish results in fewer eggs laid and ultimately a lower juvenile run than for a harvest pattern taking the same number of adults but no size selection (except possibly for populations that exceed the carrying capacity of their habitat) (Ricker 1972; Jaenicke and Celewycz 1994). Larger females also tend to dig deeper redds than smaller females, which reduces the likelihood that eggs will be destroyed by bedload movement during freshets. Larger females also select nest locations with larger gravel, which increases exchange of water and oxygen. Both of these behaviors combine to provide a greater egg-to-smolt survival (Hankin and Healey 1986; Hankin et al. 1993).

Harvest of salmonids also influences the timing of certain life history events, including adult migrations, spawning, and juvenile migrations. Frequently, fisheries are restricted to a relatively narrow window of time, particularly as stocks dwindle in numbers. Selective removal of early or late migrants can potentially result in shifts in the timing of peak migration and spawning within a population. Studies indicate that disproportionate representation of early migrants in hatchery broodstocks can cause a shift in migration timing within only a few generations (Millenbach 1973; Alexandersdottir 1987); harvesting only at the beginning or end of a run may have similar effects. Gharrett and Smoker (1993) reported that early and late-migrating adult pink salmon produce young that migrate to sea at different times. Consequently, removal of predominately early or late migrants can also alter the migration characteristics of the juvenile population.

Finally, the harvest of salmonids by humans can fundamentally alter the structure of stream ecosystems through reduction of nutrient inputs from salmon carcasses as populations decline and average size of fish decreases. Carcasses contribute significant amounts of nitrogen and phosphorus compounds to headwater streams (Cederholm and Peterson 1985; Bilby et al. 1996), the nutrients that most often limit production in oligotrophic systems. The role of carcasses in providing nutrients to stream systems is discussed in greater detail in Section 3.8.2.

6.12 Fish Introductions and Hatchery Management

Throughout history, humans have introduced fish into streams, rivers, and lakes in order to increase commercial and recreational fishing opportunities. These introductions have included both non-native species, primarily from the eastern United States and Europe, and artificially propagated native salmonids.

6.12.1 Introductions of Non-native Species

Introduction of non-native fishes into waters of the Pacific Northwest began before the turn of the century and continues today. Four primary sources of introductions include fishery management manipulations (stocking of fish); intentional introductions of gamefish by anglers; intentional or unintentional baitfish liberation by anglers; and bilge pumping of ballast water, particularly in estuaries and large rivers. Although there are few studies documenting conditions both before and after species introductions, effects of introductions on native fishes may include elimination, reduced growth and survival, and changes in community structure. For example, brown trout (*Salmo trutta*) replaced brook trout in a Minnesota stream over 15 years (Waters 1983), and cutthroat trout were replaced by more aggressive rainbow trout and brown trout in the Great Basin of western North America (Moyle and Vondracek 1985). Redside shiner were found to compete with young rainbow trout in Paul Lake, British Columbia, leading to decreased growth and survival of the young trout (Johannes and Larkin 1961). Ratliff and Howell (1992) reported that for 65 bull trout populations in Oregon considered at risk of extinction or already extinct, brook trout were the most important stressor in 26% of those populations and a contributing factor in 22%.

Moyle et al. (1986) identified six mechanisms that allow introduced fish to dominate or displace native fish including competition, predation, inhibition of reproduction, environmental modification, transfer of new parasites or diseases, and hybridization. They suggest that introduced species may thrive best where extensive environmental modification has already occurred. In the Columbia river—a system where temperature and stream velocities have been substantially altered—predator species introduced for recreational fishing, including walleye *(Stizostedion vitreum),* channel catfish (*Ictalurus punctatus*), and smallmouth bass *(Micropterus dolomieui)*, are feeding on outmigrating smolts (Palmisano et al. 1993a). Introduced grass carp (*Ctanopharyngodon idella*) and common carp (*Cyprinus carpio*) destroy beds of aquatic macrophytes, which reduces cover for juvenile fishes, destroys complex substrates that support diverse invertebrate assemblages, and increase the turbidity of water to the detriment of fishes that locate prey by sight (Moyle et al. 1986). Whirling disease—which was introduced to North American waters from Europe via shipments of frozen fish containing spores of *Myxobolus cerebralis* (Marnell 1986)—has been implicated in the decline of several important trout fisheries in the intermountain West, particularly in the upper Colorado River basin (Nehring and Walker 1996; Vincent 1996). Although this disease has been found in hatcheries within the Pacific Northwest, there is little evidence that it has affected wild trout populations in the region (Nehring and Walker 1996). Cutthroat and rainbow trout freely hybridize in the wild, with the rainbow trout phenotype becoming dominant (Behnke and Zarn 1976); similarly, the various subspecies of these trouts also interbreed.

6.12.2 Artificial Propagation of Native Salmonids

Artificial propagation of native salmonids has been used for decades as a means of replacing lost natural production resulting from various development activities and to increase returns for harvest. Hatchery programs continue to dominate expenditures of State fishery agencies in the Pacific Northwest. White et al. (1995) report that the State of Washington spent $31.3 million—35% of their total fishery budget—on salmon culture in the 1991–1992 fiscal year. Similarly, propagation of fish accounted for 42.5% of Oregon's $90.6 million budget for fisheries for the 1993–1995 biennium, whereas only 3% was devoted to management for natural production.

Although artificial propagation may in some instances increase salmon and trout available for harvest, hatchery introductions can result in a number of unintended and undesirable consequences for wild salmon and trout populations (Lichatowich and McIntyre 1987; White et al. 1995). In freshwater, interactions between hatchery and natural fishes may result in greater competition for food, habitat, or mates (Nickelson et al. 1986). Studies have suggested that carrying capacity can be exceeded during the outmigration of smolts to the ocean (Steward and Bjornn 1990). Once in the ocean, large numbers of hatchery smolts may result in density-dependent decreases in survival and growth, although evidence of density-dependent effects in ocean environments is mixed. Indications of density-dependent age and size composition have been found for various Pacific salmonids (Ricker 1981; Peterman 1987; Ishida et al. 1993). Peterman (1978) found that only a few salmonid stocks exhibit density-dependent marine survival patterns, and that these effects were predominantly within or between cohorts—little or no marine density-dependence was found between different stocks, whether derived from nearby or distant spawning areas. Holtby et al. (1990) found no evidence for density-dependent marine survival of coho salmon migrating from Carnation Creek, British Columbia. It is possible that density-dependent ocean survival may only be manifest in years of low marine productivity.

Other adverse effects of hatchery introductions include transmission of disease between hatchery and wild populations (Marnell 1986; Steward and Bjornn 1990), alterations of fish behavior, and increased predation on wild fish. In 1987, the incidence of BKD infection in hatchery spring chinook from two Snake River hatcheries was 92% to 99% (Bevan et al. 1994a). Because many fish may carry BKD for extended periods without exhibiting symptoms, cross-transmission may be substantial. The likelihood of transmission may be particularly high when fish are aggregated for transport in raceways, trucks, and barges. Alteration of behavior of wild fish, including stimulation of early migration of juveniles (Hillman and Mullen 1989), has been observed in response to hatchery introductions. Hatchery supplementation can also increase predation rates on wild stocks either directly, through predation of hatchery fish on wild fish, or indirectly by attracting predators.

In addition to ecological effects, introduction of hatchery fish may lead to genetic changes in wild populations (Hindar et al. 1991; Waples 1991a). Introduction of hatchery stocks can eliminate unique genomes in local stocks. Straying and subsequent crossbreeding may result in loss of genetic variability between populations and depressed fitness where introgression occurs. Low rates of natural straying may be beneficial in maintaining genetic variability in natural populations, but these rates may become elevated through artificial propagation (Bams 1976; Withler 1982), with potentially serious consequences for locally adapted populations.

The operation of hatchery facilities may adversely affect wild salmonid populations and their habitats in several ways (reviewed in White et al. 1995). Effluent waters from hatcheries may contain high concentrations of nutrients or disinfectant chemicals that negatively affect water quality. Disease organisms can also be introduced to streams via hatchery effluent. The construction of hatchery weirs or diversion structures impedes the migration of wild stocks and diversions of water for hatchery use reduces the amount available for wild stocks. Removal of wild fish for brood stock may threaten the genetic integrity of wild stocks, particularly for small or depleted stocks. And lastly, the removal of fish for brood stock decrease the amount of nutrients available in upstream reaches, since salmon carcasses are not deposited.

Hatchery supplementation has social repercussions that influence wild salmonids directly, as well as the ability of managers to restore salmonid populations. Hatchery supplementation increases harvest pressure on wild populations in mixed-stock and terminal fisheries ((Palmisano et al. 1993a; Lichatowich and McIntyre 1987), particularly during years when survival of hatchery fish is low due to poor environmental conditions, For example, the overcapitalization of the coho salmon fishery and subsequent overharvest of wild stocks in Oregon in the late-1970s and 1980s was stimulated in part by successful hatchery supplementation during the 1960s and early 1970s (Lichatowich and McIntyre 1987). In addition, once commercial and sport fishers have invested large sums of money in fishing boats and gear, they may become resistant to increased fishing restrictions, making it difficult for managers to enact stricter protections for wild stocks.

Finally, the long history of hatchery programs in the United States has instilled a perception in the public that habitat losses or degradation can be mitigated through artificial propagation (White et al. 1995), or that maintenance of salmon populations depends on hatcheries (Hilborn 1992). The disproportionate spending of State and Federal dollars on hatchery programs compared with protection of natural habitat and wild populations is indicative of the reliance that the public places on artificial propagation. As White et al. (1995) point out, political pressure for stocking has driven management decisions even in cases where scientific evidence has indicated stocking is not needed or detrimental. This pressure has also diverted much-needed funds from other important and more ecologically sound restoration activities.

6.13 Recreation

Although the primary influence of recreation on salmonids is fishing, there are also indirect effects related to boating, log removal, parks, and campgrounds. Stream and lake banks, riparian vegetation, and spawning redds are disturbed wherever human use is concentrated (Johnson and Carothers 1982); however, these effects are generally localized. Human concentrations at campgrounds or vacation areas may also lead to impaired water quality by elevating coliform bacteria and nutrients in streams (Aukerman and Springer 1976; Potter et al. 1984). Recreational boaters, kayakers, and rafters have less obvious, but more far-reaching effects, by removing snags from rivers and lakes. This is done for reasons of aesthetics and safety, but popular whitewater rivers and many recreational lakes are nearly devoid of snags. Removal of this wood potentially affects salmonids by reducing habitat complexity in rivers and in estuaries into which they enter. The reduced number of logs lowers estuarine and marine habitat quality for fishes just as it does for habitat in rivers (Maser and Sedell 1994).

6.14 Beaver Trapping

Other than humans, the mammal that most shaped North American waterways was probably the beaver. In pre-Columbian times, their numbers were estimated to be 4–26 km^{-2} across the United States (Naiman et al. 1986), and they provided the initial economic base for European exploration and settlement west of the Appalachians. However, because of widespread trapping in the 1800s and early 1900s, their numbers have dwindled to a fraction of their historical abundance (0.4–0.8 km^{-2} today (Naiman et al. 1986). Beavers have both negative and positive effects on water bodies and riparian ecosystems. Their feeding results in the loss of woody riparian vegetation and increased retention of fine sediments, but increases the input of large woody debris to streams. Beaver ponds increase the surface-to-volume ratio of the impounded area, thereby increasing summer temperatures. Marcus et al. (1990) suggest that in the east, temperature increases may be detrimental to trout populations, but that in the Rocky Mountains, increased temperature where waters are colder, may benefit salmonids. Beaver ponds also supplement summer low flows (Marcus et al. 1990) and provide critical over-wintering habitat for salmonids. Bank dens and channels increase erosion potential, but also offer juvenile salmonids protection from high winter flows. Beaver ponds frequently fill with sediments to become wetlands, but they retard erosion upstream and reduce sedimentation downstream. A high frequency of ponds may reduce the amount of spawning gravel through siltation (Marcus et al. 1990). When channels are once again established, these reaches produce large numbers of fish. Beaver ponds in the Rocky Mountain West were found to support larger and more numerous trout, as well as greater densities of aquatic invertebrates than undammed sections of the stream (Naiman et al. 1984). Beaver ponds may also provide a sink for nutrients from tributary streams, enhancing pond productivity, and increasing retention time (Maret et al. 1987; Naiman et al. 1986). While it is difficult to generalize about the overall effect of beaver on salmonids, Naiman et al. (1986) suggested beaver act as a keystone species "to affect ecosystem structure and dynamics far beyond their immediate requirements for food and space." Their removal has fundamentally altered aquatic ecosystem function.

7 Oceanic and Atmospheric Circulation

Until the mid-1970s, little was known about the effect of oceanic conditions on anadromous salmonids. Most research on salmonid biology focused on the freshwater environment, and fishery biologists generally attributed variation in population size to conditions in fresh water. Recent work strongly suggests that marine productivity depends on atmospheric and oceanic circulation patterns and that the abundance of salmonids and other fishes may be greatly affected by short- and long-term variation in those patterns (Mysak 1986; Roesler and Chelton 1987; Francis and Sibley 1991; Ware and Thomson 1991). Growing evidence suggests that conditions in the northeastern Pacific Ocean shifted abruptly in the mid-1970s and that salmonid populations along the entire western coast of North America have responded to these large-scale changes (Francis and Sibley 1991; Pearcy 1992). It is clear that efforts to restore freshwater habitats of salmonids need to be considered in the context of larger-scale fluctuations in numbers brought on by climatic and oceanic conditions.

In this chapter we briefly review general circulation patterns and the dominant physical processes controlling conditions in the northeastern Pacific Ocean. We then present hypothesized mechanisms by which salmonid abundance and life histories may be influenced by changing oceanic conditions. Finally, we discuss implications of long-term variability in marine conditions for strategies to restore salmonids in the Pacific Northwest.

7.1 General Ocean Circulation

Circulation in the northeastern Pacific is dominated by the behavior of the Subarctic Current and the West Wind Drift, large west-to-east surface currents situated at approximately 42–49° north latitude. These currents bifurcate as they approach North America with the Alaska Current flowing north and the California Current flowing south (Figure 7-1) (Ware and McFarlane 1989). These surface currents interact with prevailing wind patterns and the rotation of the earth to produce distinct upwelling and downwelling patterns along

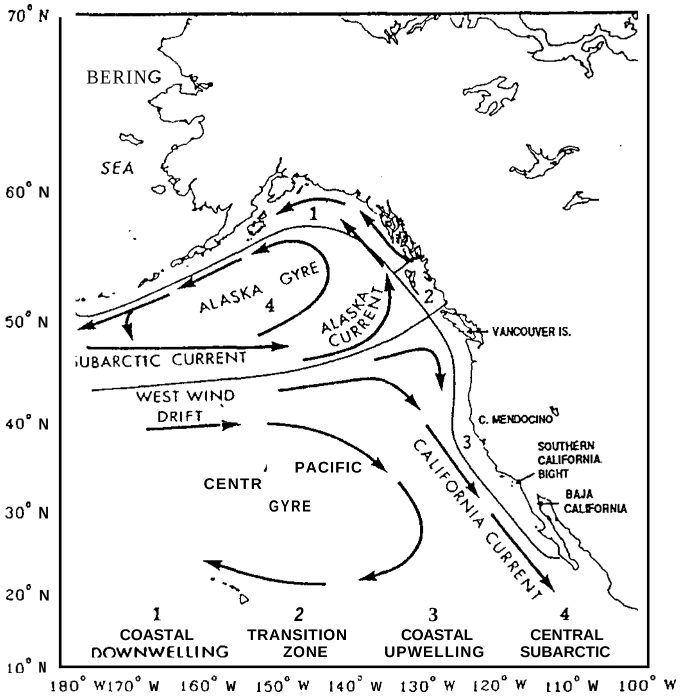

70° N
60° N
50° N
40° N
30° N
20° N
10° N
BERING
SEA
1
2
ALASKA GYRE
4
ALASKA CURRENT
VANCOUVER IS.
UBARCTIC CURRENT
WEST WIND DRIFT
C. MENDOCINO
3
SOUTHERN CALIFORNIA BIGHT
PACIFIC
CENTR
GYRE
BAJA CALIFORNIA
CALIFORNIA CURRENT
1 COASTAL DOWNWELLING
2 TRANSITION ZONE
3 COASTAL UPWELLING
4 CENTRAL SUBARCTIC
180° W
170° W
160° W
150° W
140° W
130° W
120° W
110° W
100° W

Figure 7-1. Approximate areas of oceanic domains and prevailing current directions in the northeastern Pacific Ocean. From Ware and McFarlane *(1989).* Reproduced with permission from the publisher.

different parts of the North American coast. Wind blowing across the ocean surface causes displacement of surface waters at an angle 90° to the right of the direction of the wind. South of Vancouver Island, BC, northwesterly winds generally blow along shore from May to September, causing surface waters to be transported offshore and resulting in the upwelling of cold water along the continental margin (Ware and McFarlane 1989). North of Vancouver Island, the movement of surface water is directed generally toward the shore by prevailing winds, and downwelling persists for most of the year (Ware and McFarlane 1989).

The behavior of this large-scale oceanic circulation varies from year to year and at longer time scales, depending on atmospheric conditions and particularly on the strength of the Aleutian low-pressure system off the coast of Alaska. In years when the Aleutian Low is well-developed, the position of the Subarctic Current shifts to the south (Mysak 1986), and a greater proportion of water in the Subarctic Current and the West Wind Drift is diverted northward to the Alaska Current (Pearcy 1992). These conditions are characteristic of El Niño years, when warm waters from the subtropics shift to the north. Conversely, when the Aleutian Low is poorly developed, the Subarctic Current shifts to the north, and the California Current receives a higher fraction of the total water. Ware and Thomson (1991) have proposed that long-period oscillations (40–60 years) in wind-induced upwelling significantly influence oceanic conditions along the coast. Thus, short-term variations in the strength of coastal upwelling and the occurrence of El Niño events appear to be overlaid on oscillations of longer periodicity associated with atmospheric circulation.

7.2 Ocean Conditions and Salmonid Production

Variation in oceanic circulation patterns along the North American coast greatly affects characteristics of seawater, including surface-water temperatures, salinity, sea-level height, and nutrient concentrations, which in turn affect the abundance and distribution of aquatic organisms. High temperatures, reduced upwelling, and inshore depression of the thermocline during the strong El Niño event of 1982–1983 together resulted in significant declines in phytoplankton production along the coast of Oregon (reviewed in Pearcy 1992). Similarly, zooplankton biomass was greatly reduced and species composition **shifted** to **taxa** more commonly found in more southern waters. Roesler and Chelton (1987) attributed interannual differences in the biomass of zooplankton in spring off the coast of north-central California to differences in advective transport of zooplankton from arctic waters by the California Current. Thus, changing ocean conditions can significantly affect the amount of food available to juvenile salmonids as they enter the ocean.

In addition to affecting food supply, changing oceanic conditions may also affect the distribution and abundance of predators and competitors. Holtby et al. (1990) speculated that warmer ocean temperatures off the coast of Vancouver Island may lead to northward shift in populations of large piscivorous predators such as Pacific hake (*Merluccius productus*). Increases in predator abundance and concomitant decreases in alternative prey species (e.g., Pacific herring [*Clupea pallasi*]) may result in greater mortality to salmonids off the coasts of California, Oregon, and Washington during El Niño years. Another hypothesis attributes fluctuations in ocean survival of salmonids to changes in the off-shore transport of juveniles as they enter the ocean. During years of high upwelling, smolts may be transported off shore where they are less vulnerable to sea birds and other predators that are abundant along the coast line, whereas in years of poor upwelling salmonids may remain in near-shore areas (reviewed in Pearcy 1992). Migration routes of juvenile sockeye salmon from the Fraser River differ substantially in El Niño versus La Niña years (Mysak 1986).

Regardless of the specific mechanisms controlling salmonid abundance in the ocean, the evidence is clear that oceanic conditions play a significant role in regulating survival. Numerous studies have linked marine survival of coho salmon in the ocean with the strength of upwelling (Gunsolus 1978; Nickelson 1986; Fisher and Pearcy 1988). Francis and Sibley (1991) demonstrated long-term fluctuations in the catch of coho salmon off the coast of Washington, Oregon, and California, which they attributed to changes in the marine environment caused by climatic change. Interestingly, pink salmon catches in Alaska have oscillated out of phase with coho salmon in the Oregon Production Area, indicating that conditions leading to high production of salmonids in the Coastal Upwelling Domain have adverse affects on salmon in the Coastal Downwelling Domain and vice versa (Francis and Sibley 1991).

In addition to affecting the survival and productivity of salmonids in the northeastern Pacific, variability in marine conditions has likely influenced the evolution of life history characteristics of salmonids (Holtby et al. 1989). Spence (1995) examined migration timing of coho salmon smolts from 50 populations along the coast of North America and found distinct regional differences in migration characteristics. Coho populations in the northern part of the range typically migrate during a relatively short and predictable period during the late spring. In contrast, southern populations generally exhibit a more protracted migration that peaks earlier in the spring but is more variable from year to year. Spence (1995) suggests that, in part, these differences likely reflect the adaptation of populations to differences in the degree of predictability in oceanic conditions in the northern and southern parts of the coho salmon's range. **As** knowledge of the marine ecology of salmonids increases, additional patterns in life-history characteristics of salmonids will undoubtedly emerge.

7.3 Implications for Restoration

Cycles in marine productivity have the potential to mask the effects of degradation in freshwater habitats. Lawson (1993) presented a conceptual model for considering the combined effects of oceanic cycles and habitat degradation in fresh waters (Figure 7-2). **As** freshwater habitats are degraded, salmon populations do not decline in linear fashion. Instead, a general downward trend is masked by long-term oscillations in ocean productivity. During periods of unfavorable ocean conditions, the consequences of degradation in freshwater habitats become most evident, and the risk of local population extinction becomes greatest. **As** Lawson (1993) pointed out, there may be a tendency for fishery managers and politicians to relax as populations begin to recover—which they eventually will do provided they do not go extinct during a poor ocean phase—even though the quality of freshwater habitats continues to decline. Similarly, ill-conceived restoration strategies may appear to be successful as salmonid numbers increase, even though those increases are merely the fortuitous result of improving oceanic conditions.

Long-term oscillations in ocean productivity also have a significant bearing on harvest and hatchery management. Harvest projections and limits typically are based on maximum sustained-yield models that assume a constant environment. Over the term of their prediction, these models assume linear relationships between production and yield. Such models are particularly problematic in a changing environment or in one that is tending in a direction different from that in which the model was developed. Similarly, the survival and production of hatchery fish may vary significantly with conditions at sea (Pearcy 1992). In the 1950s and 1960s coho salmon hatcheries in Oregon were enthusiastically endorsed by fishery managers and commercial fishers (who **tripled** in number over a 10-year period) because of early success that was largely the result of favorable oceanic conditions. When environmental conditions shifted in the mid-1970s, survival of hatchery coho decreased, and the overcapitalized fishery took an increasing toll on wild stocks (Pearcy 1992). Significant economic hardship for coastal communities and precipitous decline in wild coho populations resulted.

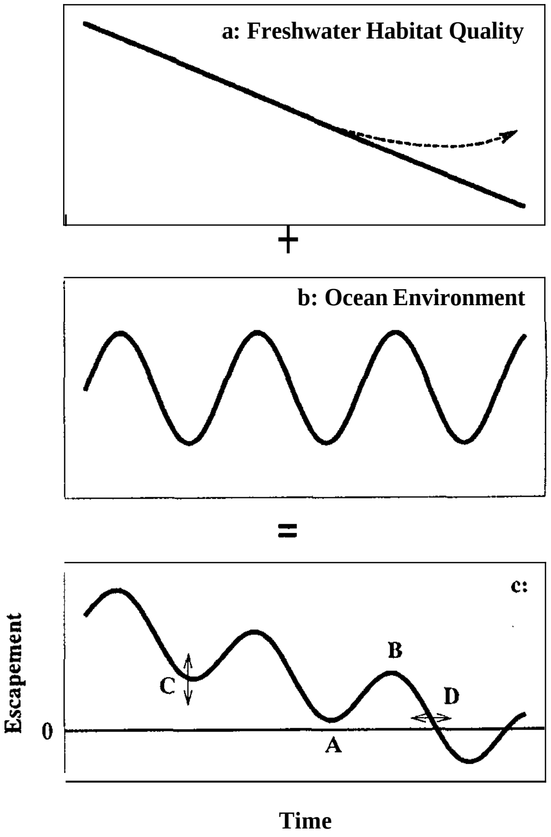


Figure 7-2. Conceptual model of effects of declining habitat quality and cyclic changes in ocean productivity on the abundance of Oregon's coastal natural coho salmon. For the labels: "a" indicates trajectory over time of habitat quality (dotted line represents possible effects of habitat restoration); "b" shows generalized time series of ocean productivity: "c" is the sum of top two panels where A = current situation, B = situation in the future, C = change in escapement from increasing or decreasing harvest, and D = change in time of extinction from increasing or decreasing harvest. From Lawson (1993). Reproduced with permission from the publisher.

Lawson (1993) concludes that, in the face of natural variation in ocean productivity, salmonid restoration should proceed in three phases: 1) short-term projects, 2) long-term projects, and 3) monitoring. Short-term projects should be directed at immediate and readily identifiable habitat problems where manipulation can temporarily enhance production (e.g., creation of off-channel pools, cleaning of gravels). Long-term projects should be directed at restoring natural ecological processes, and include such things as replanting of riparian zones or re-establishment of wetlands. Monitoring is essential to ensure that both short- and long-term projects are effective, but the metrics used to gauge success may be substantially different. Using numerical abundance of salmonids (particularly juveniles) to measure success has pitfalls within a highly fluctuating ocean environment. The focus should be whether restoration strategies are effective over decades or even centuries, not years. For long-term monitoring, indicators should measure restoration of ecological functions or processes rather than merely record fish abundance.

8 Practices to Restore and Protect Salmonids

In preceding chapters, we have discussed a wide array of effects that human activities have on aquatic salmonids, their habitats, and aquatic ecosystems in general. In this chapter, we review various practices and programs by which these effects can be avoided or substantially reduced. We first discuss practices that directly influence fish (harvest and hatcheries), then move to water- and land-use practices that directly or indirectly influence the integrity of water bodies (waterway modification, forestry, grazing, agriculture, mining, and urbanization). We conclude with a discussion of broader societal values and policies that drive resource consumption and, hence, determine the health of ecological systems. It is important to note that most programs intended to protect or restore streams have focused primarily on one problem area (e.g., stream bank, stream reach) and one stressor (e.g., grazing, agriculture, forestry, fish harvest) but have not tried to integrate other stressors into a comprehensive solution for the watershed. Although various practices and programs are discussed separately, we emphasize the need to integrate them into an ecosystem management approach at multiple spatial scales from sites and watersheds to basins and regions (see Part II of this document). In addition, regardless of the land-use activity, emphasis should be placed on preventing (rather than mitigating) damage, particularly in those areas where high-quality habitats and stable salmonid populations remain (Frissell et al. 1993; Bradbury et al. 1995).

8.1 Harvest Management

As discussed in Section 6.11, harvest of salmonids can both reduce total escapement of adult salmonids and alter the age- and size-structure of salmonid populations (Ricker 1981). Size-selective gear coupled with high rates of harvest for older age classes typically result in shifts towards younger, smaller adults. In mixed-stock fisheries where weak stocks are harvested at the same rate as more abundant stocks (including hatchery stocks), these effects can be exacerbated.

The NMFS Snake River Salmon Recovery Team recommended terminal area fishing and selective fishing as the best harvest schemes where mixed-stock fisheries include weak, depressed, or endangered stocks (Bevan et al 1994a, 1994b, 1994c). Terminal and bay fisheries (i.e., fisheries that target adults as they return to their natal streams) provide greater protection for weak stocks by targeting hatchery runs instead of wild stocks, by allowing late-maturing fish like chinook salmon to reach maturity, and by reducing the incidental mortality of subadults. Such systems are easily opened and closed to increase escapement at particular times during the run and have been employed successfully in many regions. Secondary benefits of shifting from open ocean to near-shore fisheries include reduced costs and lower risks associated with fishing.

In contrast to terminal fisheries, the management of open-ocean harvest requires a greater understanding of the distributions of different stocks in order to determine their vulnerability to the fishing fleets. The presence of stocks of differing degrees of robustness can be dealt with by managing for indicator stocks or for weak stocks. The use of indicator stocks presents the risk that the chosen indicators are not truly representative of all stocks in the fishery. Serious harm to more fragile stocks can occur when healthier indicator stocks are performing well. Conversely, potential harvest from healthy stocks may be lost by managing strictly for the weakest stocks (Kope 1992; Restrepo et al. 1992). Weak stock management has the latter risk (higher levels of yield foregone) and the added difficulty that smaller populations tend to have less data available concerning their production and recruitment mechanisms, as well as fishery impacts (Peterman 1978; Peterman and Steer 1981; Lestelle and Gilbertson 1993). In practice, a mixture of the two techniques is usually attempted, managing for indicator stocks or complexes of stocks while also including attention to weaker stocks vulnerable to the fishery.

Harvest methods can also be changed to target hatchery stocks and reduce incidental mortality of wild populations. Wild fish caught in traps and fish wheels or by hook and line (especially with lures or barbless hooks) are less likely to die than those captured with gillnets or by trolling. Special sport-angling restrictions, including catch-and-release angling, minimum size or slot limits, and bag limits may further reduce mortality or minimize size-selective harvest. Accurate monitoring of escapement levels of specific stocks is essential for establishing exploitation levels that ensure the long-term persistence of individual stocks (e.g., Cooney and Jacobs 1994).

8.2 Hatchery Practices

Growing evidence of adverse ecological, genetic, and social consequences of hatchery operations (reviewed in Section 6.12) has prompted many fishery scientists to recommend substantial modification, curtailment, or elimination of hatchery programs for salmonids (Lichatowich and McIntyre 1987; Waples 1991b; Hilborn 1992; Meffe 1992; Bevan et al. 1994a; White et al. 1995). Reductions in hatchery production would disrupt harvest levels over the short term. Nevertheless, competition, predation, disease transmission, and genetic introgression resulting from fish stocking threaten the persistence of wild salmonids, particularly those populations that have been depleted because of habitat degradation and overharvest.

White et al. (1995) recommended that fish stocking be limited to three kinds of temporary programs and two types of prolonged programs. They suggested that temporary stocking should be used only to 1) recolonize native species into waters where fish have been extirpated by human activities, after those activities (e.g., habitat destruction, introduction of non-native species, overharvest) have been reduced or eliminated; 2) create populations in new artificial waters (e.g., constructed lakes or ponds), and 3) sustain a presently overharvested fishery through a planned program of downsizing and transition to other employment or from reliance on hatchery fish to reliance on wild fish. Prolonged programs they recommended include hatcheries to 1) augment weak stocks (put-and-grow stocking) in waters having little or no reproductive habitat but substantial productive potential, but only where it will not harm indigenous biota and 2) provide highly artificial opportunities for recreation where fishing will be intensive (e.g., put-and-take stocking of catchable-sized fish, for quick and easy catch in urban ponds). White et al. (1995) argue that supplementation stocking (defined as increasing natural production) is "out of keeping with ecological reality and rational management." Although certain societal values may be achieved with hatcheries (Schramm and Mudrak 1994), these values can often be attained through angling restrictions and restoration of wild populations without hatchery supplementation.

The issue of hatchery supplementation is particularly contentious as it relates to augmenting threatened and endangered stocks through the use of wild brood stock from the endangered population. In deciding whether to use artificial propagation to conserve endangered species, a key consideration is the likelihood that such efforts will actually benefit the listed species (Waples 1991b). Although artificial propagation of Pacific salmon has been carried out on a large scale for many decades, almost all these efforts have been directed at enhancing fisheries harvest. Attempts to increase natural production through the use of artificial propagation is a relatively recent enterprise that has, to date, produced mixed results (Miller et al. 1990). Removal of threatened and endangered species for use as brood stock may increase the genetic vulnerability of the population and hence its long-term prospects for survival. For example, a recent modeling exercise by Currens and Busack (1995) indicated that the risk of extinction for a chinook salmon population was greater with supplementation than without it. Consequently, the use of artificial propagation to conserve listed species should be viewed as experimental and highly risky given the historical effects of hatcheries on reducing the biological diversity of salmonids (Bottom 1997).

One of the most disruptive influences of hatcheries has been the introduction of non-native species to Pacific Northwest waters. These species include competitors, such as brook trout and American shad, as well as several large piscivores including walleye, smallmouth bass, largemouth bass (*Micropterus salmoides*), and striped bass (*Morone saxatilis*). Obviously, the most direct way to minimize these impacts is to cease stocking both waters that contain wild salmonids with non-native species and waters from which non-natives can disperse into salmonid-bearing streams. Other management strategies that encourage elimination of non-native fishes include direct removal by piscicides and electrofishing, and indirect removal through use of unrestricted catch limits (Bevan et al. 1994a). Restoring streams and rivers to their natural temperature and flow regimes may reduce the spread of non-native species into salmonid streams. For example, evidence from California suggests that streams with natural flow regimes are less prone to invasion by non-native species than highly regulated streams where seasonal flow fluctuations have been artificially dampened (Baltz and Moyle 1993). Finally, it is important to note that many State and Federal agencies continue to promote the stocking of non-native species into certain waters, although new introductions have been curtailed in recent years as understanding of the ecological consequences of species introductions has increased. Stocking programs both contribute directly to declines in native salmonids (and other native biota) and create social constraints that may impede restoration of wild fish (e.g., sport angling constituencies that favor fisheries for non-native species).

8.3 Waterway Modification

Throughout human history, waterways have been used for water supply, waste discharge, power generation, and transportation—values that have collectively made waterways centers for development. With growth in human populations and commerce, many streams, rivers, and estuaries were increasingly simplified by dams, channelization, revetments, snagging, and removal of wetlands and side channels. Seasonal peak and low flows were stabilized by dams, although in systems with hydroelectric facilities, daily fluctuations were increased to meet power demands. Water quality deteriorated as a result of these changes combined with discharge of municipal and industrial wastes, as well as water withdrawals for industrial, agricultural, and domestic uses. Particularly damaging to aquatic life were increased temperature, turbidity, sedimentation, toxics, nutrients, and oxygen-demanding wastes. The progressive commercial, agricultural, and urban development stimulated additional alterations, often at increasingly greater distances. In places such as western Europe, where these changes have been taking place for centuries, natural channels and high water quality remain in only the most remote areas.

Substantial strides have been made toward improving the water quality of the Nation's surface waters. Many of the most extreme instances of pollution have been substantially reduced. Sensitive forms of aquatic life and recreational opportunities are returning to all but the worst of our water bodies, and the frequency of waterborne disease has been dramatically reduced. Nationally, these changes were stimulated and financed by the Federal Water Pollution Control Act (FWPCA) and subsequent amendments that form the Clean Water Act (CWA; see Chapter 9); however, several progressive States, as well as industries, preceded the FWPCA in cleaning up point-source discharges. Several lessons in waterway protection and restoration can be learned from the Federal water quality legislation, nonfederal responses, and related legislation: 1) a clear set of goals and objectives is needed; 2) funding must be set aside for applied research and training; 3) cost-sharing grants are often necessary to construct treatment works; 4) numerical criteria are required for various pollutants; 5) both site-specific and basin-wide permitting, monitoring, and reporting systems are useful for controlling pollution and evaluating compliance; 6) improved land-use can effectively control diffuse pollution; 7) ambient biological criteria and biological surveys are necessary for evaluating the biological effectiveness of controls; 8) reducing wastes frequently saves money for those who discharge pollutants; 9) the Federal government can provide overall guidance and direction, but States can effectively conduct the monitoring and enforce the regulations; 10) the effectiveness of pollution-reduction programs ultimately depends on the character and creativity of those discharging contaminants into rivers and streams.

Recently, researchers have identified the vital role of channel complexity, riparian zones, and floodplains in the productivity and diversity of the aquatic habitat and its organisms (Sedell and Luchessa 1982; Maser et al. 1988; Gregory et al. 1991; NRC 1992). Numerous programs are presently underway to increase river-floodplain interactions for the benefit of instream habitat and riparian zones. Approaches range from active natural channel restoration to nonstructural passive methods for floodplain protection. The Kissimmee River, Florida, restoration project has been called a model for watershed restoration (Niering and Allen 1995). It calls for physical reconfiguration of the river to re-establish backwaters and contact with the floodplains and for re-establishment of historical inflows from the river's upper watershed. A similar effort has been planned for portions of the Missouri River (Hesse and Sheets 1993; Galat and Rasmussen 1995). That approach includes an evaluation of predisturbance conditions with the intent of identifying areas where side channels, wetlands, and bottomland hardwood forests can be restored through the removal of revetments and set back of levees.

Other researchers have identified strategies to minimize adverse affects of channelization through stream renovation. McConnell et al. (1980) suggested that snagging only one side of a stream halves costs and deleterious impacts to the environment. Nunnally et al. (1978) proposed minimizing erosion and reducing hydraulic efficiency by maintaining natural meanders when channelizing streams. The more natural stream channels and riparian vegetation also improve biological and aesthetic benefits. Alternative strategies such as these involve significant environmental tradeoffs but are less destructive than traditional channelization.

Several proposals have been put forth to restore natural ecological processes in rivers by removing existing dams. The Maine Legislature requested that Edwards Dam be removed from the Kennebec River to improve migration of Atlantic salmon. Despite fish-passage modifications, the dam still blocks migrations of salmon as well as migrating sturgeon, shad, and smelt while providing electricity to fewer than 2,000 homes (NRC 1992). The Elwha River Ecosystem and Fisheries Restoration Act (Public Law 102-495) was passed in 1992 to remove two dams on the Elwha system for the purpose of restoring the riverine ecosystem and historic runs of five Pacific salmon species, once among the most prolific runs on the Olympic peninsula of Washington (Wunderlich et al. 1994). The dams also are associated with the disappearance of 22 bird and mammal species from that area of Olympic National Park with consequent, annual recreational revenue losses of $500,000. Electricity from the dams is sold to a paper company, and replacement sources of power have been identified. An analysis determined that removal of both dams is the only option that will allow full restoration of the watershed (NRC 1992). In Oregon, two dams on the Rogue River have been identified for potential removal. The Bureau of Reclamation (BR 1995) estimated that removal of Savage Rapids Dam would increase escapement of adult salmon and steelhead by 26,700 fish, increase harvest by 87,900 fish, and cost taxpayers nearly $8,000,000 less than dam retention (with modifications for improved fish passage); these benefits to fisheries would be attained while still providing irrigative withdrawals, the primary function of the dam. Additional dam removals are called for throughout the Pacific Northwest because of inadequate or no fish passage, excessive sedimentation, structural deterioration, and hazardous or unsafe conditions.

A variety of programs have been aimed at maintaining the viability of fish stocks by mitigating the impacts of dams. These programs provide upstream passage for mature fish and downstream passage of smolts. The Northwest Power Planning Council's Strategy for Salmon identifies immediate actions to aid juvenile passage (NPPC 1992a, 1992b): screen all turbines, improve bypass systems, lower reservoir pools during smolt migration, barge past dams, place a bounty on predators, and boost flows during out-migration. The Snake River Recovery Team made similar final recommendations to NMFS (Bevan et al. 1994a, 1994b, 1994c).

Reintroduction of beaver has also been recommended as a means of restoring stream habitats. Beaver dams can increase habitat complexity, slow stream incision, and increase flows during the dry season by allowing greater subsurface storage of water in floodplain areas. Although beaver ponds may eliminate spawning areas, warm the water, and reduce dissolved oxygen, they create additional areas for rearing, over-wintering, escaping freshets, and trapping sediments. Their introduction to higher gradient salmonid streams is associated with higher salmon densities and significantly greater overwinter survival (Phillips 1987; Swanston 1991).

The Fish and Wildlife Service has developed the Instream Flow Incremental Methodology (IFIM) to provide a means of systematically evaluating alternative flows for the protection or enhancement of aquatic resources in regulated river systems (Armour and Taylor 1991). The IFIM couples models of physical habitat conditions at different stream discharges with information on specific habitat requirements of fish to estimate minimum instream flows needed below river impoundments. Maintaining instream flows is especially important for eastside streams where irrigation needs or hydroelectric operations may severely deplete base flows. Though widely used, a number of concerns of IFIM concepts have been expressed. Nestler (1990) summarized these concerns, arguing that depth, velocity, and substrate are inadequate estimates of habitat quality and that habitat quality is only a coarse estimate of population density. Stalnaker (1990) argued that IFIM has been used to establish minimum flows that are frequently violated and that ignore other necessary flows such as floods. Armour and Taylor (1991) pointed out that the methodology was designed to evaluate alternate flow regimes not as the definitive answer for flow disputes. The results simply provide a framework for negotiating flows to be maintained.

During the past two decades, increasing effort and resources have been committed to instream artificial structures intended to improve fish habitat. The National Research Council (1992) provided a summary of 22 habitat-improvement evaluations deemed successful based on increased fish density during the period evaluated. No time period for evaluation is given, nor are metrics provided that were used as the basis for the evaluation. The results from these studies should be interpreted with caution because they originated in systems east of the Rocky Mountains, systems having different climate, geology, sediment transport, hydrology, and gradient than Northwest streams. In contrast, Frissell and Nawa (1992) surveyed artificial structures in streams of western Oregon and Washington and concluded that "commonly prescribed structural modifications often are inappropriate and counterproductive." They reported frequent damage to artificial structures, particularly those located in low-gradient reaches and in streams with recent watershed disturbance. When evaluated for 5–10 year damage rates, overall median failure rate was found to be 14% and median damage rate (impairment plus failure) was 60%. They concluded that streams with high or elevated sediment loads, high peak flows, or highly erodible bank materials are not good candidates for structural modifications. Beschta et al. (1991) surveyed 16 stream-restoration projects in eastern Oregon and found that instream structures frequently had negative effects on aquatic habitats (e.g., altered natural biotic and fluvial processes), were inappropriate for the ecological setting (e.g., boulders or large wood placed in meadow systems that historically never had such structures), or did not address the full suite of riparian functions that contribute to habitat quality. They concluded that in most instances instream structures are unwarranted and should be eliminated as a restoration method; re-establishment of riparian vegetation through corridor fencing or rest from grazing was determined to be far more effective in restoring habitats. Restoration of fourth order and larger alluvial valley streams, areas identified as having the greatest potential for fish production in the Pacific Northwest, will require natural watershed and riparian processes to be re-established over the long term. Reeves et al. (1991) described numerous structure and habitat manipulations (gravel cleaning, gabions, weirs, log sills, cedar baffles, fishways, boulders, log structures), and provided an evaluation of their use and applicability for variable life history requirements and differing watershed settings. They cautioned that much work has been done with very little pre- and post-evaluation of the results, and that successful future projects will depend upon careful evaluation of existing projects. Reeves et al. (1991) concluded that 1) habitat rehabilitation should not be viewed as a substitute for habitat protection, 2) prevention of initial habitat degradation is more economical of total resources than repairing that degradation, and 3) some damage to streams is simply irreversible.

8.4 Forestry Practices

As linkages between upland and riparian forest management and aquatic ecosystems have become better understood, a gradual evolution toward more ecologically sound forest practices has occurred. Each of the States in the Pacific Northwest has adopted forest practice rules that provide greater protection to riparian areas along fish-bearing streams, and certain damaging practices (e.g., splash damming) have been eliminated altogether. In this section, we briefly review methods for minimizing the effects of forest practices on aquatic ecosystems. More information is provided in this section than in those for other land uses both because of the substantial overlap between forested lands and salmonid distributions and because the literature relating forest practices to salmonid habitats is far more extensive.

The effects of forest practices on watershed processes and aquatic ecosystems are influenced by the harvest schedule, methods, equipment, and unit location as well as by the site preparation methods, intermediate treatments, and the road location, construction, and maintenance. To minimize impacts, practices can be selected that are least disruptive to natural watershed processes. Forest practices that will most effectively protect stream ecosystems vary with local physical and biological characteristics.

8.4.1 Upland Forest Management

Although riparian activities pose the greatest risk to salmonids, upland practices affect surface erosion, mass wasting, hydrologic processes, and nutrient dynamics; therefore, these must also be considered.

Silvicultural Systems

Rotation schedule influences watershed dynamics by determining the frequency of disturbance to the watershed, the total area disturbed, and the quantity of materials delivered to a water body through time. Most commercial forests in western Washington, Oregon, and California have been harvested on a rotation of 45–100 years (Frissell 1991; Hicks et al. 1991a), though shorter rotations are becoming more common. Harvest rotations are typically longer for forests east of the Cascade Crest because climatic conditions lead to slower growth rates of trees.

Effects of harvest on stream temperatures, hydrology, surface erosion, and the probability of mass failures are generally greatest in the years immediately following logging because the degree of devegetation and soil disturbance is highest during this period. Lengthening the harvest rotation decreases the time that the landscape is disturbed and reduces the probability of catastrophic events. For example, if the risk of landslides is increased for a period of 15 years after logging, then a stand managed on a 60-year rotation will be vulnerable for 25 years per century versus 15 years for a stand managed on a 100-year rotation (Frissell 1991). On a regional scale, if the average rotation is 60 years then 25% of the landscape is vulnerable to landslides at any time, versus 15% with a 100-year rotation or 5% with a 300-year rotation.

Harvest can generally be divided into "even-aged" and "uneven-aged" methods (Young and Giese 1990). Even-aged methods are those in which the timber stand consists of trees of similar age and size and include such methods as clear-cutting, seed tree methods, and shelterwood cuts. Uneven-aged methods consist of those where trees are selectively harvested and where the resulting stand consists of trees of varied ages.

Clear-cutting has been the dominant harvest method in forests of the Pacific Northwest since the turn of the century (FEMAT 1993). Clear-cutting is potentially more disruptive of natural watershed processes—including hydrology, sediment transport, energy transfer, nutrient cycling, and stream habitat development—than other methods because virtually all vegetation is removed and soil is usually highly disturbed.

Patch cuts are relatively small clear-cuts distributed over the landscape. The effectiveness of patch cutting in mitigating effects on watershed processes depends on the size and location of the harvest units and the percentage of the watershed harvested. Small patches may reduce hydrologic impacts because vegetation in areas surrounding the cut may take up some of the additional available water. However, compared with a single large clear-cut, multiple patch cuts may facilitate greater snow deposition, resulting in higher runoff during rain-on-snow events. While several patch cuts may be less disruptive to soil and hydrologic processes than a single clear-cut of equivalent total area, the resulting fragmentation of the watershed may be more detrimental. Dispersing timber harvest units and roads increases the probability of multiple chronic landslides across the landscape, rather than concentrating disturbances in particular watersheds. Also, for some wildlife species, large, intact systems function better as habitats than highly fragmented systems. Furthermore, multiple patch cuts may require more road mileage than a single harvested area, further diminishing the ability of habitats to support certain wildlife species, These tradeoffs should be considered when developing timber harvest strategies.

Seed tree and shelterwood cuts differ from clear-cuts in that some trees are left on site to provide seed sources for regeneration and, in the case of shelterwood cuts, to provide some shade for seedlings. Following the establishment of seedlings, the remaining large trees are removed, leaving an even-aged stand. Both of these methods can potentially reduce hydrologic effects and surface erosion. Trees that remain on site may exhibit compensatory growth in response to increased water availability, thereby minimizing increases in runoff. Standing vegetation also serves to reduce erosion, although seed trees are likely to have minimal effect on the probability of mass failures.

Selective harvest methods entail the removal of only a portion of the merchantable trees from a cut. Traditionally, only the largest trees were removed. Alternatively, harvest may involve selective removal of younger trees while leaving large, older trees standing. New forestry or "structural retention" techniques (sensu Swanson and Berg 1991) involve selecting a set of windfirm dominant and codominant trees to be retained for a rotation or more, thereby restoring large-diameter trees that are typically absent from most industrially managed forest lands. New forestry practices may also include retention of snags and downed wood, maintaining all size and age-classes of trees, and leaving patches as refugia. The intent of all of these techniques is to hasten the development of the characteristics and diversity typical of old-growth and late-successional forests that are desirable to some forms of wildlife, including the northern spotted owl and marbled murrelet. However, it is not known whether such measures will benefit other late-successional forest species (FEMAT 1993). The benefits of selective harvest vary depending upon the percentage of the basal area removed **and** the composition of the remaining stand. If properly done, selective harvest can maintain stream shading and input of allochthonous materials, minimize disturbance to soils, reduce soil compaction by ground-based equipment, and minimize effect on hydrologic processes. Disadvantages of selective harvest may include increased frequency of disturbance, suboptimal regeneration of trees, increased density of roads, and increases in harvest costs under some circumstances.

Harvest (Yarding) Systems

Considerable disturbance to soils occurs during skidding or yarding of logs from the stumps to landings. Several systems have been devised for yarding that differ in their impact to soils. Ground-based operations using tracked or rubber-tired skidders generally result in the greatest disturbance to soils both in terms of degree of compaction and the percentage of total area affected. Such disturbances increase surface runoff and erosion potential. Because most compaction occurs with the first few passes of heavy equipment, soil disturbance can be reduced by establishing designated skid trails, thereby minimizing the area receiving traffic. Because compaction is highest when soils are moist, impacts may be further reduced by skidding either during the dry season or when the ground is frozen.

In cable-yarding systems, logs are attached to a cable and dragged to the landing. In general, a lower percentage of the logged area is disturbed with this method; however, skid marks may channel water, thereby facilitating erosion. High lead systems are similar, with the cable running through an elevated pulley. Skyline systems lift part of the log off the ground as it is dragged to the landing site, further reducing the impact to soils. Full suspension methods lift logs completely off the ground. Helicopter logging has also been employed to reduce the need for roads and ground-based equipment. In general, the area affected by logging equipment and percentage of bare soil remaining is greatest for tractor and cable logging, intermediate for highlead systems, and least for full suspension, skyline, and helicopter logging (Pritchett and Fisher 1987).

Site Preparation

Regeneration of coniferous vegetation in some regions involves reducing shrubs and deciduous trees, eliminating logging debris, and preparing soils for planting. Three techniques have been used for site preparation: burning of slash, mechanical clearing of vegetation, and chemical treatments. As noted in

Section 3.3, the effects of fire on soil characteristics vary with the intensity of the burn. High-intensity fires can eliminate litter layers and create hydrophobic conditions in surface soils, thereby increasing the amount of exposed soil and the potential for surface runoff. Low-intensity burns, on the other hand, generally cause minimal damage to soils (Pritchett and Fisher 1987), and are therefore less likely to result in changes to hydrologic or erosional processes.

Mechanical clearing potentially has the greatest effect on soil conditions and hydrologic processes, and often causes greater damage to soils than the initial logging and yarding. Because heavy equipment must travel over most or all of a site, the potential for soil compaction and exposure of bare soils is high, which can lead to increased surface runoff and erosion (Pritchett and Fisher 1987). In addition, valuable topsoil may be redistributed, with much of it ending up in burn piles.

Chemical treatments may be least physically damaging to a logged site; however, care must be taken to ensure that chemicals do not reach stream systems. Limiting spray operations to calm days, using the minimum effective concentrations, and refraining from spraying in riparian zones can minimize the risk of exposure of aquatic organisms to toxic chemicals. In addition, applications can be timed so as not to overlap with sensitive life-history stages of fish or other aquatic biota. Norris et al. (1991) concluded that the herbicides used in forestry are relatively immobile in soils and that leaching into subsurface waters is less likely in forested soils than in other environments. From a hydrologic standpoint, application of herbicides is likely to extend the period of increased water yield from a site by reducing evapotranspiration losses. Conversely, at sites where deciduous vegetation will replace coniferous trees if untreated, chemical treatments may prevent reductions in summer streamflow that may occur over longer periods of time.

intermediate Treatments

Intermediate treatments are actions designed to enhance tree growth, and include thinning, pruning, and fertilization. The impacts of thinning and pruning activities are generally related to the type of equipment used and the care demonstrated by the operator. Lighter equipment will generally cause less disturbance to soils and ground cover than heavy equipment. In a well planned thinning, the use of equipment will be restricted to designated roads and skid trails, thereby minimizing additional soil compaction. Indiscriminate use of equipment, on the other hand, can result in compaction of soils that were left intact during the original harvest operation. Provided that additional soil compaction does not occur, thinning or pruning is unlikely to have a significant effect on water balance in a forest stand because remaining vegetation will take up additional water that becomes available. The effects of fertilization on aquatic systems can be minimized by refraining from fertilizing riparian zones. Fertilizing in ephemeral channels can lead to high concentrations of nitrogen in downstream areas when rainfall begins in the fall and fertilizers are mobilized (Norris et al. 1991). Thus, foresters should avoid applying fertilizers near all permanent streams, ephemeral streams, and drainage channels.

Road Construction and Maintenance

Logging roads are primary sources of sediments to streams, both through chronic erosion and as trigger points of mass failures. In addition, the higher the road density within a watershed, the greater the probability of significant alteration to hydrologic processes. Impacts can be substantially reduced by careful placement, construction, and maintenance of roads. Furniss et al. (1991) provided a thorough discussion of ways for minimizing the effects of road construction on aquatic ecosystems. Table 8-1 summarizes their recommendations, and much of the information in this section comes from their review. In addition, the forest practice rules of most western States contain guidelines for minimizing impacts of forest roads.

Minimizing total roaded area and careful placement of roads are the most fundamental means for reducing sediment inputs to channels. Long-range planning of road systems within a watershed helps minimize total surface area of roads and can reduce construction costs as well. In general, roads should be located away from stream channels, particularly in steep terrain where the likelihood of fill material washing into the stream is high. However, it may be preferable to construct roads in valley bottoms rather than on slopes that have a high probability of failure. In these instances, a buffer strip between the road and stream can reduce disturbances to the channel. Roads located too near the active channel in unconstrained reaches can impede the natural lateral migration of the stream channel across the floodplain, disrupting natural processes of erosion and deposition. Locating roads on ridgelines, as opposed to mid-slope areas, and on dry soils instead of in wet areas, also minimizes erosion risks (Furniss et al. 1991). Knowledge of local soils and geology is essential to ensure wise placement of roads. Hummocky ground, jack-straw trees, and sag ponds are often good indicators of unstable hillslopes subject to slumping or slides. Creek crossings are frequently the sites of significant erosion. Minimizing the number of crossings is both desirable and required under forest practice rules of some States

Table 8-1. Recommendations for minimizing impacts of forest roads on aquatic habitats. Recommendations based on Furniss et al. (1991).

Impact	Recommendation
Drainage	Disperse drainage rather than concentrating it.
	Avoid discharging large amounts of water into non-drainage areas.
	Avoid altering natural drainage patterns by means of water bars or culverts.
	Use outslope drainage to disperse runoff.
	Where inslope-and-ditch drainage is used, relieve the ditchline of drainage at frequent intervals.
	Use discharge pipes to route water away from fill slopes.
	Design drainage structures to accommodate peak streamflow based on at least 50-year-interval flood.
	Control scouring at culvert outlets with energy dissipators such as heavy rock, rip-rap, or other materials.
Stream crossings	Avoid channel width changes and protect stream banks with rip-rap or other retaining structures.
	Use retaining walls to reduce excavation near stream channels.
	Design road to approach creek crossings at right angles.
	Design crossings so they will not divert water down the road.
	Install instream culverts at angles and heights that allow fish passage. Culverts should be placed below original stream bed and gradient should be less than 1%.
Road beds, cut and fill slopes	Use minimum design standards for road width, radius, and gradient.
	Minimize excavation by using natural features.
	Design cut slopes to be as steep as practical.
	Where practical, surface roads to control erosion.
	Remove earth material and debris from streambanks to prevent them from being washed into the stream.
	Restrict gravel extraction to areas above high-water level of design flood.
	Do not incorporate organic materials into road fills.
	Use end hauling rather than side casting on steep slopes to minimize risk of fill-slope failures.
	Minimize height of cut slopes to reduce risk of failure.
Road location	Avoid mid-slope locations in favor of higher, flatter areas (ridgetops).
	Do not locate roads within inner valley gorge.
	Avoid slopes with excessive wetness.
	Avoid slopes requiring large cut and fill areas.
	Locate roads to minimize roadway drainage area and to avoid modifying the natural drainage areas of small streams.
	For valley-bottom roads, provide a buffer strip of natural vegetation between the road and stream.
	Locate roads to take advantage of natural log-landing areas.
	Minimize the number of stream crossings.
	Locate stream crossings to minimize channel changes.

(e.g., California). When crossings are unavoidable, they should be located where the amount of channel modification and fill material is minimized (Furniss al. 1991).

The principal considerations in designing and constructing roads to minimize effects on salmonid habitats are ensuring adequate drainage, preventing excessive sedimentation, and providing for fish passage at stream crossings. Sediment transport is generally tightly coupled with the routing of water on the landscape. Thus, most sound design and construction techniques are devoted to maintaining natural drainage patterns, preventing the concentration of runoff, avoiding discharge of water onto unstable fill slopes, and designing structures to accommodate extreme hydrologic events (Table 8-1).

Regular and timely maintenance of logging roads helps ensure that drainage and erosion control structures are functioning properly and allows identification of problems that could have adverse consequences. The costs associated with maintenance are generally low compared with reconstruction costs after a significant failure. Grading roads to ensure outsloping surfaces, and clearing of drainage ditches and culverts can ensure that drainage occurs as intended. Where problems are observed, installation of additional ditch-relief culverts or large culverts may alleviate erosion and drainage problems. Seasonal road closures may also be an effective way to reduce sediment delivery to streams.

Once harvesting has been completed at a site and the road is no longer needed, reseeding of the road bed with grasses reduces the amount of exposed soil and thereby decreases surface erosion. However, this practice has little effect on the potential for deep mass wasting. Recently, there has been growing support for revegetating or decommissioning roads by pulling sidecast material back onto the road bed and reforming the natural slope (Harr and Nichols 1993). Decommissioning of roads involves disturbance to restore natural morphology. Nevertheless, Harr and Nichols (1993) reported that decommissioned roads and landings sustained little or no damage following two significant rain-on-snow events that caused substantial damage to main haul roads in northwest Washington. Similarly, Weaver et al. (1987) reported that obliteration of problem road surfaces and fills, deconstruction of stream crossings, and re-contouring of disturbed slopes were effective techniques for reducing sediment input to streams in Redwood National Park. Other techniques aimed at surface and rill erosion problems were less successful at reducing sediment delivery and tended to be more costly. Both Harr and Nichols (1993) and Weaver et al. (1987) noted that a careful survey of road conditions that allowed them to identify significant problem areas was essential to successful cost-effective application of rehabilitation techniques.

8.4.2. Riparian Forest Management

Floodplain and riparian forests in the Pacific Northwest once supported some of the largest and fastest growing trees, and were among the first forests that were logged because of the relative ease of transporting logs via waterways. Recent research, however, has recognized the importance of floodplains, floodplain wetlands, and riparian zones for storing and slowing floodwaters, absorbing pollutants from runoff, reducing sediment delivery to streams, maintaining channel complexity, supplying shade and large woody debris, providing shallow water areas for feeding and spawning fish, and supporting a highly diverse community of plants and animals (BLM et al. 1994; Cederholm 1994). The practice of leaving riparian buffer strips along streams is now widely applied and is viewed as perhaps the most important aspect of protecting stream habitats from the effects of logging and other land-use activities (Cummins et al. 1994).

Three important considerations in establishing buffer zones are: 1) the width of the buffer zone, 2) the level of activity allowed within the riparian zone, and 3) whether riparian buffers are needed for tributary streams that do not contain salmonids. Appropriate buffer widths are the topic of much debate and a number of alternative approaches for determining adequate buffer widths have been proposed (FEMAT 1993; Cederholm 1994; FS and BLM 1994a; Cummins et al. 1994). The appropriate width of buffer zones depends on the specific functions that are being considered. Figure 3-2 illustrates generalized curves for the zones of influence of riparian vegetation relative to key riparian functions (FEMAT 1993). Litter inputs and bank stability are generally provided by trees within 0.5 potential tree heights of the channel. Shading and large woody debris are provided by trees farther from the stream channel; in some instances, significant amounts of large wood may be carried to the channel in landslides or debris flows originating outside of the riparian zone. The effect of vegetation on sediment and nutrient inputs may extend even farther from the channel, though these influences are more difficult to define. Complete protection of salmonid habitats requires that all of these functions be maintained. A more thorough discussion of riparian management practices, including State riparian protection rules, is presented in Part II, Section 14.2.3 of this document.

The influence of riparian vegetation also depends on physical and biological characteristics of the specific location, including topography, soil type, geology, and vegetative cover. For example, the likelihood that large logs will end up in the stream channel is greater on steep slopes compared to gentle slopes. The effectiveness of riparian vegetation as a sediment filter depends on slope and soil type. Topographic shading may reduce the importance of shading by vegetation in some locations. The FEMAT approach to riparian buffers establishes buffer widths based on stream and land classifications; these widths can be adjusted if it is demonstrated, through watershed analysis, that riparian conservation objectives will not be compromised. This approach acknowledges that critical instream characteristics can be maintained with variable buffer widths determined in accordance with site conditions.

The second significant consideration in the management of riparian areas is the level or intensity of disturbance allowed within them. State forest practice rules generally allow harvesting of timber within riparian areas but at levels lower than in surrounding uplands, which are commonly clear-cut. Depending on the State, specific criteria establish the number of trees, species composition, basal area, overstory and understory canopy cover, or other measures of the vegetation to be left within the riparian zone (reviewed in Section 14.2.3). Deviations from standards (both more and less conservative) may also be granted based on site inspection or production potential. The implicit assumption of these rules is that some level of disturbance within the riparian zone is acceptable and will have minimal affect on salmonids and their habitats. For example, under Washington's forest practice rules, shade retention requirements for temperature control vary depending on whether he stream is classified as sensitive; thus, incremental temperature increases may be allowed that, while not causing direct physiological stress to aquatic organisms, may influence ecological interactions (see Chapter 4). An alternative view is that the target of riparian management should be no impairment of riparian function and that downstream and cumulative effects must be considered. Cederholm (1994) proposed that riparian zones should be identified and buffer zones should be established around the riparian zone to prevent modification of riparian function.

Where riparian zones have already been altered by human activity, the long-term prospects for recovery of large conifers may be limited without active manipulation of riparian vegetation. For example, many riparian zones in coastal forests have been converted to dense, alder-dominated stands that leave little opportunity for conifer regeneration. In these instance, thinnings can be used to create openings that allow sufficient light for conifer re-establishment. For such activities, leaving heavy equipment outside of the riparian zone can ensure that soils and streambanks are not disturbed.

Finally, impacts of logging can be reduced if buffer zones are left around small headwater streams that themselves do not support salmonids (Cumins et al. 1994). In particular, steep headwater drainages are frequently the trigger points of landslides. Minimizing road construction and logging around first order and temporary channels can prevent frequent mass soil movements that propagate downstream, to the detriment of salmonids.

8.5 Range Practices

Livestock grazing occurs on approximately 70% of both Federal and nonfederal lands in the West (GAO 1988b; Fleischner 1994), making it the most widespread land use in the region. Many wildlife refuges, wilderness areas, and even some national parks (e.g., Great Basin National Park) are grazed by domestic livestock. Since the 1930's, rangelands in the Pacific Northwest have benefited from less intensive grazing; however, the majority of western rangelands remain in deteriorated condition. Although thorough surveys of range condition on both private and public lands are lacking, the available evidence indicates that between 50% and 65% of rangelands are in poor or fair condition (Busby 1979; GAO 1991; Heady and Child 1994). An even higher percentage of western riparian areas are in degraded condition. On BLM and Forest Service lands in Colorado, Arizona, Nevada, and Oregon, between 60% and 93% of riparian areas are considered to be in poor or fair condition (GAO 1988a; Armour et al. 1994). Poor upland conditions increase sediment loads and alter hydrologic regimes, leading to channel incision, channel widening, and further deterioration of riparian zones. Similarly, damaged riparian areas are unable to buffer streams from changes brought on by degradation of upland areas. Thus, restoring salmonid habitats in rangelands requires improving livestock management in both upland and riparian areas.

8.5.1 Upland Range Management

Although strategies for improving salmonid habitats in rangeland streams have to date focused primarily on modifying grazing practices within the riparian zone, effects of grazing on hydrologic and sediment transport processes necessitate improvement of range practices in upland areas as well. Hydrologic changes occur in response to loss of vegetation or change in plant species composition, as well as to changes in soil permeability brought on by reduced organic content, splash erosion, and trampling by livestock. Similarly, sediment transport processes are linked to vegetation cover and the routing of water from the hillslope to the stream. Consequently, restoring the natural rate of water and sediment delivery streams can be attained by restoring more natural vegetation assemblages in uplands.

The retention of adequate vegetative cover in uplands is critical to minimizing hydrologic and sediment impacts and can be achieved through a variety of means including 1) lowering livestock stocking rates; 2) controlling animal distribution through fencing, herding, salting, and watering; 3) changing species composition of livestock; and 4) altering the timing of grazing (Heady and Child 1994). Each of these practices can influence the percent of forage that is utilized, the composition of vegetation that remains on site after grazing has ended, and the degree of soil disturbance.

Reducing stocking rates not only provides greater protection to streams and watershed, it can result in improved condition and value of livestock as well. At high stocking rates livestock do not gain weight quickly or at all, or they lose weight and condition. At the same time, the range deteriorates or fails to recover, costing ranchers money over the long term. In addition, high stocking rates maximize financial losses when livestock prices fall between the time of calf acquisition and sale. Yield rises constantly with increased livestock density (assuming a constant environment) towards some optimum, then falls sharply slightly beyond that optimum production level. This makes it very difficult to select an optimum stocking rate in a predictable environment, let alone a highly variable one. Heady and Child (1994) report that, for both environmental and financial reasons, increasing numbers of managers are shifting toward lower stocking levels.

Livestock tend to concentrate in areas near water, shade, preferred vegetation, salt, and on relatively level topography. As a result, these heavily used areas may become overgrazed and trampled, leading to erosion and hydrologic disruption. Development of alternative water sources, salting, fencing, and herding can all be used to ensure more uniform utilization of forage (and hence remaining cover) and to reduce impacts associated with soil compaction.

Although common in eastern and southern Africa, where native diseases limit domestic livestock, game ranching and cropping have not been widely adopted in this country. Livestock grazing benefits wildlife species that prefer habitats altered by livestock, but harms those species preferring natural habitats, or those competing with livestock for food (Heady and Child 1994). Therefore, depending on the desired species of wildlife, livestock may need to be reduced or removed. Native wildlife populations can sustain high harvest levels; for example, some deer populations have been harvested at rates of 33%–50% annually for decades without detrimental effects (McCullough 1979; Heady and Child 1994). Various game management and harvest programs are also possible on these lands, depending on whether the objectives are trophies, meat, or wildlife viewing. Selling prices range from several thousand dollars for a trophy animal to a few dollars for wildlife viewing. However, wildlife viewing is a nonconsumptive, repeatable activity. Benefits in addition to range improvement include lower management costs, leaner meat supply, and higher biological integrity.

Because plant responses to defoliation differ depending upon the season of grazing, vegetation can be protected by restricting grazing during certain times of the year. Four general types of grazing plans have been utilized: continuous, repeated seasonal, deferred, and rest (Heady and Child 1994). As the name implies, continuous grazing entails grazing throughout the growing season and usually some part of the dormant period, thus the length of time varies with climate. Repeated seasonal grazing refers to annually grazing the same pasture during a specific season, similar to the patterns of migratory wildlife. Deferred grazing means no grazing is conducted until key plants have completed reproduction. Rest once referred to a year without grazing but has since been generalized to any specified period. Where range condition is less than excellent or in arid and semiarid regions, range recovery may require many years of deferment or rest, The success of each is a function of site characteristics, periodic monitoring, and low stocking densities. These will determine which pastures to graze, which season or year to graze them, and for how long.

8.5.2 Riparian Range Management

Western riparian areas are among the most productive ecosystems in North America, yet their present condition is believed to be the worst in American history, largely because of livestock grazing (Fleischner 1994). Acknowledging the need to manage the entire watershed, Barrett et al. (1993) established a BLM goal of 75% or more properly functioning riparian wetlands by 1997. They defined proper functioning condition as adequate vegetation, landform, or large woody debris to 1) dissipate stream energy associated with high flows, thereby reducing erosion and improving water quality; 2) filter sediment, capture bedload, and aid floodplain development; 3) improve floodwater retention and groundwater recharge; 4) develop root masses that stabilize streambanks against cutting action; 5) develop diverse ponding and channel characteristics to provide the habitat, water depth, flow duration, and stream temperature necessary for fish production, waterfowl breeding, and other uses; and 6) support greater biodiversity.

Heady and Child (1994), Platts (1991), Chaney et al. (1990), and Kauffman and Krueger (1984) identified numerous options to be considered singly or in combination for achieving grazing goals while maintaining or improving fish habitat. They include resting from grazing, controlling livestock distribution, controlling livestock numbers, controlling forage use, controlling timing of forage use, grazing the type of livestock best suited for a given area, and artificially rehabilitating stream riparian ecosystems. Elmore (1992) and Platts (1991) provide greater elaboration on specific grazing strategies and their influence on riparian ecosystems and fish habitats (Table 8-2).

Grazing stress and trampling impacts are a function of how and when a given strategy is used and not simply on the total number of animals grazing at a particular site. Strategies that include corridor fencing, rest rotation with seasonal preference, and complete rest from grazing provide the greatest potential for rehabilitating degraded riparian areas (Platts 1991). Other grazing strategies may be effective under specific circumstances, depending on local climate, hydrologic conditions, soils, stream character, and plant species composition (Chaney et al. 1990; Elmore 1992). Selective timing for rotation grazing and strategies that allow growth of riparian vegetation during critical periods may be just as effective as reducing grazing intensity in some areas (Elmore 1992). Seasonal grazing strategies can also minimize trampling impacts to streambanks (Kauffman and Krueger 1984; Chaney et al. 1990). Having a clear set of riparian objectives and understanding vegetation potential at the site are critical to identifying effective riparian grazing strategies. It is also important to note that rangeland streams have unique attributes that may make them more vulnerable to anthropogenic stresses. For example, rangeland streams typically experience low or highly fluctuating flows and rocky or highly alkaline soils that severely limit riparian vegetation (Crouse and Kindschy 1981). If other preexisting stress levels are high, the capacity to absorb additional grazing stresses or to recover when they are removed may be low. Thus, effective grazing strategies must integrate the natural potential and the expected grazing stress for a given stream reach (Elmore 1992). In the absence of site specific information, deferment of grazing until riparian vegetation has returned to pre-grazed conditions remains the safest course of action.

Numerous case histories document the effectiveness of various grazing strategies in improving riparian conditions. The GAO (1988b) and Chaney et al. (1990) recently reviewed riparian restoration efforts on BLM and Forest Service lands and reported substantial improvement in riparian and stream conditions in many instances. Although riparian fencing and reductions in stocking rates generally proved to be most effective measures, deferred grazing, rest-rotation, daily herding, and development of off-stream water sources also yielded positive results. Ecological benefits noted included improvements in riparian grasses and shrubs, regeneration of cottonwood trees, increases in summer streamflow, reductions in summer water temperatures, narrowing and deepening of stream channels, increased pool area, increased stability of streambanks, improved substrates, and increased fish density.

In their review, Kauffman and Krueger (1984) cited studies indicating that one-to-two years of rest out of three provided improved riparian vegetation as long as forage consumption was below 60%–65%.They also noted that although rest-rotation strategies may improve condition of vegetation, increases in trail formation and trampling may cause streambank erosion and instability. Finally, they note that daily herding of sheep from stream bottoms improves utilization of upland forage while providing greater protection to riparian areas.

As in uplands, improvements in grazing practices within riparian areas can provide economic benefits to ranchers. The GAO (1988a) report cites several instances where animal unit months increased following recovery of riparian vegetation. Livestock excluded from riparian zones were forced to graze on underutilized upland vegetation, resulting in healthier, heavier livestock.

Frequently, riparian restoration efforts on rangelands have involved both changes in grazing practices and placement of instream structures. Kauffman and Krueger (1984) reported that better livestock management was a less costly strategy than instream structures for restoring channel integrity. They added that the need for instream structures is negated by rest from grazing and that structures are often ineffective when not accompanied by modification or cessation of grazing within the riparian zone. Similar conclusions were reached by GAO (1988a), Chaney et al. (1990), and Beschta et al. (1991). Instream structures were considered by Heady and Child (1994) to treat the symptoms of improper grazing rather than the causes. Thus, instream structures are likely unnecessary for rangeland stream restoration except under unusual circumstances, and they should not be used as a substitute for improved management of livestock.

8.6 Agricultural Practices

Agriculture and human settlement began on floodplains because of the availability of water and the fertility of soil. As a consequence, floodplains

Table 8-2. Evaluation of the effects of various grazing strategies on riparian habitats. From Platts (1991). Used with permission from the publisher.

Strategy*	Level of vegetation use in riparian area	Control of animal distribution	Stream-bank stability	Brushy species condition	Seasonal plant regrowth	Riparian rehabilitation potential	Rating†
Continuous season-long use (C)	Heavy	Poor	Poor	Poor	Poor	Poor	1
Holding (S or C)	Heavy	Excellent	Poor	Poor	Fair	Poor	1
Short-duration, high intensity (C)	Heavy	Excellent	Poor	Poor	Poor	Poor	1
Three-herd, four pasture (C)	Heavy to moderate	Good	Poor	Poor	Poor	Poor	2
Holistic (C or S)	Heavy to light	Good	Poor to good	Poor	Good	Poor to excellent	2 – 9
Deferred (C)	Heavy to moderate	Fair	Poor	Poor	Fair	Fair	3
Seasonal suitability (C)	Heavy	Good	Poor	Poor	Fair	Fair	3
Deferred rotation (C)	Heavy to moderate	Good	Fair	Fair	Fair	Fair	4
Stuttered deferred rotation (C)	Heavy to moderate	Good	Fair	Fair	Fair	Fair	4
Winter (S or C)	Heavy to moderate	Fair	Good	Fair	Fair to good	Good	5
Rest rotation (C)	Heavy to moderate	Good	Fair to good	Fair	Fair to good	Fair	5
Double rest rotation (C)	Moderate	Good	Good	Fair	Good	Good	6
Seasonal riparian preference (C or S)	Moderate to light	Good	Good	Good	Fair	Fair	6
Riparian pasture (C or S)	As prescribed	Good	Good	Good	Good	Good	a
Corridor fencing (C)	None	Excellent	Good to excellent	Excellent	Good to excellent	Excellent	9
Rest rotation with seasonal preference (S)	Light	Good	Good to excellent	Good to excellent	Good	Excellent	9
Rest or closure (C or S)	None	Excellent	Excellent	Excellent	Excellent	Excellent	10

* C = cattle, S = sheep.

† Strategies are rated on a scale of 1 (poorly compatible with fishery needs) to 10 (highly compatible).

and riparian areas in agricultural lands remain among the most disturbed areas in the landscape, particularly because agriculture most often involves complete replacement of natural vegetation and repeated disturbance of soils through tillage. Regulations, management practices, and any other activities that completely protect floodplains, riparian areas, and uplands in a natural state ensure that channel and riparian functions are unimpaired. Recognizing that totally protecting large numbers of agricultural watersheds or floodplains is impractical under current demands for food and other crops, we focus in this section on management strategies that preserve most critical functions while allowing continued use of agricultural lands. These strategies are aimed at conserving water and soil, protecting and restoring riparian vegetation, and minimizing use of chemical pesticides and fertilizers.

8.6.1 Upland Cropland Management

Current agricultural practices offer ample opportunities for conservation. Of all the water diverted and consumed in the Pacific Northwest, approximately 90% is used to irrigate crops (Wilkinson 1992), and this percentage is even higher in several sub-basins east of the Cascade Crest (Muckleston 1993). Irrigation and associated water quality problems are the major stressor in reduced salmon runs in the Yakima, Walla Walla, and Umatilla Rivers (NPPC 1986). Where irrigation withdrawals are substantial, one of the most important current management actions for restoring salmonids is maintaining adequate year-round instream flows. Without sufficient flows, other restoration activities are likely to be ineffective. Instream flows can be maintained through a combination of instream water rights and water conservation efforts. State and Federal fish and wildlife agencies have established most water rights, but landowners in some States, including Oregon, have also granted or sold water rights to fishermen's groups or the State, generating more net income than they did from marginal crops (Wilkinson 1992). One of the potentially most useful tools for maintaining or restoring instream water is the 1908 U.S. Supreme Court decision on *Winters vs. United States,* which decreed that Indian tribes possessed water rights that were superior to those established by State law. Another legal approach involves the common-law public trust doctrine, which holds that the rights to water on larger water bodies cannot be controlled by a single part of the population (Wilkinson 1992) and that private property owners are prohibited from acting in a manner inconsistent with public trust interests (Johnson and Paschal 1995).

Water conservation is the most fundamental way to provide more water for aquatic life. Environmental concerns surrounding the construction of new dams on salmonid-bearing streams underscore the need to improve the efficiency with which water is used in agriculture. Water use can be decreased by timing irrigation to coincide with periods (both daily and seasonal) of low solar radiation to minimize evapotranspiration losses. Installation of cement-lined canals can reduce transpiration losses by noncrop plants that typically grow in unlined irrigation ditches. Use of drip or trickle irrigation systems instead of flood irrigation minimizes evaporation losses as well as reducing the need for weed control. Conversion of water-intensive crops such as rice and alfalfa to more drought-resistant crops can also minimize water use. Leveling of fields with laser technology has been employed to minimize runoff of irrigation water from croplands. Each of these methods maximize the efficiency with which water is used while simultaneously reducing chemical and thermal pollution associated with irrigation return flows. Effective regulatory tools for water conservation include monitoring and taxing water use, and using graduated pricing for the water consumed (Wilkinson 1992).

Screening of irrigation canals and pump intakes is also essential to protect salmonids. An investigation of 225 intakes along the Columbia River during the late 1970s indicated that 70% lacked proper screening (Swan et al. 1980); subsequent surveys concluded that 30% of intakes remained improperly screened after irrigators were notified of inadequacies (Swan et al. 1981). In many instances, existing screens may be sufficient to protect outmigrating smolts but insufficient to prevent entrainment of smaller fry (Palmisano et al. 1993a).

Control of sedimentation from agricultural lands remains a significant concern in many lowland streams in the Pacific Northwest. Nationwide, five of the six most popular soil conservation programs funded by the Agricultural Stabilization and Conservation Service in recent decades involved increased vegetative cover (ASCS 1992). In southeast Washington, replacement of row crops and small grains with permanent vegetation or hay and pasture was predicted to reduce erosion rates to 0.1%–0.01% of their former level. In the same study, various forms of conservation tillage reduced soil erosion by 13%–95%, depending on precipitation (SCS et al. 1984). Grassed waterways continue to be popular as a means to limit soil erosion and many enlightened farmers leave riparian buffers along surface waters. Dairy farms typically have wet weather controls for limiting run-off from manure heaps and cattle are fenced from stream access. Currie (1994) recommended greater use of existing incentives and disincentives. Among incentives, he included greater tax abatements through the Washington Open Space

Program, and higher Federal subsidies for implementing best management plans. At the same time, Currie proposed that farms not implementing best management plans be subjected to reduced Open Space exemptions and higher fines for farm pollution. Federal laws also encourage farmers to take highly erodible lands out of production by making farmers that cultivate such lands ineligible for Federal price supports, crop insurance, loans, or disaster payments (see Chapter 9). Watershed analysis, including risk assessment, management plans, and monitoring, is also proposed as a successful tool that could be adapted from forestland management to farmlands (Currie 1994).

Reducing the use of chemical pesticides and fertilizers is another important conservation strategy for agricultural lands. Organic farming and integrated pest management are also growing in popularity on small agricultural operations. Demand for biocide-free crops continues to rise along with concerns about the human health and ecological effects of pesticides. In addition, the cost of biocides and problems with neighboring landowners and contaminated ground water has restricted their use in some areas. These changes have mostly occurred on small farms that can more effectively implement integrated pest management. Where chemicals are used, contamination of streams can be minimized by applying chemicals at their minimum effective concentration, by spraying during periods of low wind, and by maintaining no-spray riparian buffers.

8.6.2 Riparian Cropland Management

As in forest and rangeland management, the practice of leaving riparian buffer strips is central to conservation of streams and rivers in agricultural lands. Vegetated buffer strips greatly reduce the delivery of sediment and chemical pollutants from croplands. In addition, riparian buffers stabilize streambanks, provide shade, and contribute large wood to streams that frequently lack these attributes. Riparian forests, together with fencerows, frequently constitute important wildlife habitats in agricultural landscapes otherwise devoid of suitable habitats.

Also important to the restoration of streams and rivers in agricultural lands is the re-establishment of natural floods. Just as it is now acknowledged that large woody debris is critical for maintaining channel complexity, it is also apparent that floods are necessary to prevent channels from incising, redistribute coarse sediments, build floodplains, introduce large wood, and propagate natural riparian vegetation. Traditional State and local floodplain zoning and easements allow some measure of control over the type of activities that can occur on floodplains. This approach may be quite successful if those allowed activities cause only minimal disruption of the floodplain ecosystem (e.g., natural parks, fishing access points). Typically, however, floodplain zoning restricts only those activities that incur extensive damage during floods (e.g., structures), but allows other activities (e.g., logging, grazing, farming) that significantly change the characteristics of floodplain ecosystems and, hence, the functions they perform (Kusler 1979).

An example of a more protective approach is the Banner Drainage and Levee District in Illinois, which is being restored to lakes and wetlands (NRC 1992). Another approach is the Willamette River greenway in western Oregon that includes 255 river miles and includes sloughs and side channels (NRC 1992); however, agricultural lands are exempt from greenway regulations and in many reaches there is no natural floodplain or only a narrow strip of native vegetation. Oregon also has a program to provide tax relief to landowners that maintain natural riparian zones. Federal laws have sought to protect wetlands in agricultural areas by eliminating U. S. Department of Agriculture benefits (e.g., price and income supports) for areas where farmers cleared and drained wetlands for crop production (see Section 9.4). Water Quality 2000 (1992) and NRC (1992) both stressed the importance of protecting and restoring existing wetlands rather than trying to recreate them after they have been converted. Recognizing the ecological need for riparian and wetland areas to flood and the substantial cost to humans when they do, it may be more prudent to relocate activities from floodplains than to subsidize continued development of floodplains through channel maintenance, dam and levee construction, Federal flood insurance, and "disaster" relief (NRC 1992). The benefits include saving money, allowing natural processes to re-establish habitat, and reducing hazards to human residents.

8.7 Mining Practices

In 1872, hard rock mining was encouraged through legislation by the Federal government so that miners could easily obtain mining claims, produce metals, and settle the West. Increasing concerns for the ecological costs of mining, particularly pollution of streams and rivers by mining wastes, has prompted calls for more strict regulation of mining activities. Wilkinson (1992) proposed several policy options including banning patenting of Federal lands (obtaining ownership by simply paying $2.50–$5.00 per acre and investing $100 per year), setting strict reclamation requirements (including liability bonds), charging value-based royalties on minerals removed, and prohibiting mining in sensitive areas. He also recommended leasing, rather than selling, the land to miners and evaluating whether there is a net public benefit of the mining. The Western Division of the

American Fisheries Society (WDAFS 1994), in calling for reform of the 1872 mining law, recommended these same measures, as well as public participation in all aspects of mining regulation, programs for monitoring compliance, time limits on Plan of Operation approvals, and other environmental safeguards. Nelson et al. (1991) added that riparian and stream enhancement should also be part of the reclamation process.

8.7.7 Upland Mining Practices

Reclamation of mining sites typically focuses on preventing mine-generated solid wastes and toxic materials from entering waters. This can be most effectively achieved by restoring natural landscape contours, followed by re-establishment of vegetation. To allow restoration of natural vegetation on mined lands, it is critical that topsoil be set aside before mining begins (Nelson et al. 1991). Toxic materials should be buried below the root zone to prevent uptake of by plants. In addition, toxic wastes should be buried away from areas where leachates are likely to enter streams or groundwater. Ground water does not pass through these materials to streams or the water table. When the area is returned to its natural contours (this can be problematic when the volume of spoils exceeds that of the original ore), the soil is replaced and revegetated with the original flora or acceptable substitutes. Revegetation may require seeding or introduction of vegetative propagules, as well as tilling, mulching and fertilization. It is critical that lands be stabilized as soon as possible to limit erosion. Mining-generated solids and seepage or runoff from mines should be kept from streams by proper planning and control structures such as erosion barriers and lined ponds.

8.7.2 Riparian and Instream Mining Practices

Mining in or near streams requires additional precautions to those for uplands (Nelson et al. 1991). Effluents may be treated with hydrated lime or sulfite and then aerated to raise the pH and allow the metals to precipitate. Reverse osmosis and electrochemical precipitation are also effective following acid neutralization. If the channel form and substrate have been altered, a channel and riparian zone should be developed that allows normal ecological processes to occur. Nelson et al. (1991) stressed that such channels are not static; they should resemble the pre-existing channel in their bed, banks, riparian vegetation, and flows. Re-establishment of riparian woody plants may require transplanting.

Instream and floodplain aggregate mining poses special problems since, by its nature, this activity involves disturbance to channel morphology. In reviewing effects of gravel extraction on streams in Oregon, OWRRI (1995) made several recommendations for minimizing degradation to salmonid habitats: 1) prohibit, regulate, or otherwise manage small operations (i.e., less than 50 cubic yards); 2) conduct gravel removal in streams in a manner to minimize potential impacts on salmonid habitats; 3) allow gravel removal by bar skimming only under restricted conditions (i.e., where the gravel bar is not an active spawning, rearing, or feeding are for salmonids; where adequate gravel recruitment exists to replenish the bar; where berms and buffers can be used to control streamflow away from the excavation site; where gravel can be removed from above the low-water level during low flows; and where the final grading of the bar does not significantly alter the flow characteristics of the river at high flows); 4) restrict deep-water dredging for gravel production to areas where it is presently practiced; 5) do not allow a net loss of wetlands for all fill/removal operations; and 6) use biological streambank stabilization methods where possible. Secondly, they stressed the need for monitoring and research to evaluate impacts, improved database capabilities and use for the managing agency, implementation of GIS-based resource management, and allocation of sufficient funds to monitor resource abundance, conditions, and use. From a policy standpoint, they suggested that 1) the burden of proof of "no significant impact" should be shifted to permit applicants where proposed activities are expected to result in significant direct or indirect impacts to salmonids; 2) gravel extraction should not be allowed in reaches of ODSL-managed streams that support sensitive, threatened or endangered species; and 3) gravel extraction should not be allowed from reaches of ODSL-managed streams that are part of Aquatic Diversity Areas or that support source salmon populations. Finally, they concluded that gravel removal operations may provide potential opportunities for increasing salmonid habitats through reconnecting former floodplain gravel pits to riverine systems; using gravel mining as a potential method for creating wetlands, off-stream channels, lakes and ponds, or salmonid spawning beds; and using gravel mining to improve spawning areas by improving sediment quality, increasing channel sinuosity in streams that have been channelized or otherwise simplified. In all of these instances, active restoration should be ecologically based and carefully studied prior to implementation.

8.8 Urban Practices

Restoring and protecting salmonid habitats in urban areas is one of the most difficult challenges facing land managers because many disturbances to the urban landscape are essentially irreversible, barring a radical change in social values.

Urbanization fundamentally alters water quality in streams (Smart et al. 1985). The high percentage of impervious surfaces leads to increased runoff, making stream hydrographs much flashier. For example, a 20% increase in impermeable surfaces can double runoff in a storm event (Birch et al. 1992). Channel morphology is modified by intentional conversion of natural drainage channels into conduits and as a consequence of decreased channel stability resulting from higher peak flows. Riparian vegetation is also extensively modified, with gallery forests and shrubs being converted into buildings, roads, parking lots, and lawns. Protecting and restoring lakes and streams from the effects of urbanization, therefore, involve reducing the areal extent of urbanization, removing pollutants from the waste stream, and conserving natural channels (Wanielista 1978).

Increasingly, land-use planning is used to restrict urban development from most sensitive areas, although the effects of unplanned (with respect to aquatic resource concerns) development persist. Relatively successful examples of such planning include the California Coastal Commission, the Oregon Land Conservation and Development Commission, and the Tahoe Regional Planning Agency (Wilkinson 1992). The impact of urbanization also can be reduced by favoring high-density housing, by greater utilization of bicycles and mass transit, and by placing major transportation networks underground: in other words by designing cities for people instead of automobiles (Doxiadis 1971). Not only do these more rational city designs decrease the amount of impervious surfaces, but also they decrease the amount of pollutants collected by and discharged from roads. As with other land uses, rigorous basin planning is incorporated in successful urban planning (Birch et al. 1992).

The major way of removing urban pollutants from the waste stream is by industrial and sewage treatment plants. Most **U.S.** cities now have secondary sewage treatment, and many industries have tertiary or secondary treatment. Secondary treatment, however, only transforms wastes into nutrients that are then discharged into rivers. As water purification and waste treatment costs rise and growing human populations increase the demand for limited water, more households and municipalities begin water rationing and recycling. Household rationing has taken the form of xeriscaping in place of lawns, limiting lawn irrigation, smaller toilet tanks, low-flush toilets, composting toilets, rinse-only showers, and low-discharge shower nozzles (Wilkinson 1992). The major opportunity for domestic recycling is in the reuse of washwater on gardens and lawns and in toilets (Wagner 1971). Municipal rationing involves higher and progressive water and sewer rates as well as lawn-watering restrictions. Cities also recycle sewage water for irrigation or find it less expensive to purify chemically treated effluent for reuse than to withdraw lower quality water from rivers. Both approaches are in use elsewhere in this country (Wagner 1971) and in Europe. Industrial rationing and recycling have become more common as discharge permits became more restrictive.

Another substantial source of polluted waters is stormwater runoff from lawns, roofs, parking lots, and streets. These sources can be reduced by decreasing their surface areas as discussed above and by requiring point-source discharge permits on outfalls. More typical practices can be broken into nonstructural and structural approaches (Wanielista 1978). Nonstructural approaches include street cleaning (especially mechanical broom and vacuum sweeping), cleaning of catch basins, dust control, restrictions on the use of lawn chemicals, erosion control at construction sites, and the use of wetland systems (vegetated floodplains, marshes, ponds riparian zones) as natural filters. Structural management practices for reducing stormwater runoff problems include retention basins, constructed wetlands, land injection, rooftop and parking lot storage, and sediment traps. Illicit connections to storm drains are very common in cities (Birch et al. 1992). Improper connections can be located by associating chemicals in the effluent with likely producers, dye studies, and TV inspections. Birch et al. (1992) provide a thorough set of guidelines for controlling erosion and sedimentation from construction sites, including matting and mulching open soil, erosion barriers, sediment traps, interceptors and drains on cut-fill slopes, and removal of sediment from roads. Monitoring and maintaining control structures at such sites, especially during storms, is important.

8.9 Regional Planning and Management Efforts

Most management efforts for protecting and restoring salmonid populations have focused on the fish (harvest restrictions, fish passage, hatchery supplementation) or on aquatic habitats (water quality criteria, physical habitat structure, flow) within a limited area dictated by the particular land- or water-use activity. Although population- and site-specific efforts are an essential component of salmonid restoration, many issues related to long-term persistence of salmonids involve larger spatial scales and hence require statewide or multistate planning.

FEMAT (1993), PACFISH (FS and BLM 1994a), and INFISH (FS 1995) are examples of coordinated, Federal, land-use planning that, despite what some perceive to be compromises and limitations, represents significant progress towards regional conservation. Several States have begun similar efforts. For example, Oregon's statewide land-use planning law is designed to protect forest, agricultural, and coastal lands from urbanization. Goal 5 of that law requires conservation and protection of lands needed for fish and wildlife habitats, water areas, wetlands, watersheds, and groundwater. Although it is statewide in scope, it is implemented and monitored at the county level with little statewide assessment of status or trends by the Department of Land Conservation and Development. Oregon's riparian set aside law, overseen by the Department of Fish and Wildlife, offers landowners tax deductions for protecting such areas. In response to Senate Bill 1125, Oregon Department of Forestry (ODF 1994) recently developed rules providing increased riparian protection for all fish-bearing forest streams. Levels of protection vary with water body use, type, and size. A higher design and maintenance standard for new stream-crossing structures was also promulgated. Aquatic diversity areas (Henjum et al. 1994), similar to FEMAT's key watersheds, have been mapped by the Oregon Chapter of the American Fisheries Society for the entire State. These areas, together with locations of unusually high salmonid production, have been incorporated into a framework for allocating salmon restoration funds (Bradbury et al. 1995). Protection and restoration of such areas throughout the region are necessary to preserve and expand salmonid populations that can support sustainable harvests.

The other States in the region have developed similar planning and management systems. For example, Washington Department of Natural Resources has rules for riparian protection that vary with water body use, type, and size. In addition, Washington promotes watershed analysis as a means of identifying sensitive and high-risk areas within watersheds, or to minimize disturbances to aquatic ecosystems resulting from forest practices. California's Coastal Zone Management Act restricts development on sensitive coastal and estuarine areas. All four States in the region have water quality (temperature and dissolved oxygen) standards for the protection of salmonids. Generally these standards would be protective if monitored and enforced. In addition, the States have local zoning laws restricting building types and densities, The water quality and land-use standards, however, differ from State to State and lack a statewide planning and monitoring design, let alone a regional one.

8.10 Individual and Social Practices

Direct alteration of habitat by humans remains the single greatest threat to both terrestrial and aquatic biodiversity (Noss 1992). Most habitat alterations affecting salmonids relate to resource consumption of some sort—the use of water, electricity, wood and wood products, meat and wool, food and nonfood crops, and mineral resources. Per capita consumption of resources is an order of magnitude greater in the United States than it is in much of the world. Therefore, each of us can minimize our indirect effects on salmonids by markedly reducing consumption of all resources. There are a number of things that we can do individually and as a society to begin these changes'and reduce our environmental impact.

8.10.1 Short-term Individual and Governmental Actions

The amount of water available for aquatic life in streams and lakes can be increased by reducing the amount diverted for domestic, industrial, and agricultural uses. Water conservation begins at home in how we shower, launder, flush toilets, landscape, irrigate, and use electricity. Those same functions of cleaning, cooling, waste disposal, irrigation, and power consumption offer opportunities for water conservation in industry and agriculture as well. Potential for increased efficiency in these water uses has been demonstrated by various voluntary and mandatory water conservation measures implemented during recent droughts in California, Oregon, and Washington, Key aspects in conserving water and electricity are accurate monitoring of uses, internalizing environmental costs associated with water use (e.g., dam impacts, hatchery operations, wastewater treatment), and progressive pricing so that greater use results in proportionately higher rates.

Wiser use and conservation of metals, particularly aluminum and heavy metals, would also reduce the demand for hydropower and hence the adverse effects of dams on anadromous salmonids. Over 40% of the aluminum used in this country is produced in the Pacific Northwest, and fully 20% of the total energy sold by BPA is used by aluminum smelters and other energy-intensive industrial processes. Excessive packaging results in enormous waste of aluminum. Every three months, Americans discard enough aluminum to rebuild the Nation's entire commercial air fleet. Recycling aluminum requires approximately 5% as much energy as refining the metal from bauxite, Because aluminum is an important component of many car parts, demand for aluminum can also be curtailed through reduced auto use and ownership, as well as greater reliance on mass transit and other forms of transportation.

As with aluminum, wood products have considerable conservation potential. Worldwide, humans used over 30% more wood per person in

1991 than in 1950, mostly as fuel, but in the developed countries per capita wood consumption has been declining for most of the century (Durning 1994). Other trends are less encouraging. The United States produces 26% of the world's industrial wood with Russia a distant second. Average house size in the United States has increased from 100 m^2 in 1949 to 185 m^2 in 1993. This represents 50%–100% more space per person than West Europeans and Japanese, respectively. World paper consumption has increased 20-fold since 1913, mostly in the highly industrialized nations. In 1960, the average family in the United States spent $500 per year on packaging, and the Nation as a whole paid $190 million for junk mail (Packard 1960). Currently, over half a million trees are used each week to print this Nation's Sunday newspapers, much of which consist of advertisements that many readers discard and that promote consumption of unneeded products. Demand for wood products can be reduced by creating "paperless" offices, decreasing packaging, recycling paper, and developing alternative sources of fiber. Recent development of chipboard has increased fiber supply options to include wood waste, previously undesirable weed trees, and agricultural wastes such as straw and hemp. Agricultural waste fibers and weed trees also hold promise for paper manufacturing. Current methods to reduce waste and increase recycling and manufacturing efficiency could halve present wood consumption in the United States (Postel 1994).

Because livestock production and commercial fish harvest have substantial effects on salmonids, it is useful to examine ways to reduce consumption of beef and fish. Alternative protein sources, such as grains and legumes, would reduce the demand for salmon harvest and the need for range-fed livestock. Only 10% of the protein ingested by cattle is converted to tissue; consequently, beef is a relatively inefficient source of protein for humans compared with grains and legumes. The growing number of vegetarians (currently estimated as 4%) in the United States and the continuing popularity of wildlife hunting as sources of protein are both desirable trends, as is reduced meat consumption in general. Diets low in red meat reduce the risk of death and disease, in addition to indirectly benefitting salmonid habitats.

Development of alternative energy sources could reduce dependence on hydropower and potentially allow for the removal of some hydroelectric dams. Energy conservation is a major source of new energy, but wind farms have considerable potential near the coast and in the Columbia Gorge. In addition, solar power and fuel-cell units in individual buildings are likely to become more popular as their unit costs decrease and hydropower rates increase.

Perhaps one of the most effective ways in which our culture could conserve salmonids and their environments is to remove many direct and indirect subsidies that encourage resource use and consumption. Many of these subsidies were initially intended to facilitate the development of the West, long before the environment was a significant societal concern, and they continue at substantial economic and environmental expense. For example, postal customers subsidize both the delivery of junk mail they do not want and its disposal in landfills. Taxpayers indirectly pay for building in high risk areas (floodplains, faults, fire-prone lands, ocean shores) through costs of fire suppression and disaster relief. Farmers are aided by taxpayers through drought and crop insurance and Federal price supports. In the West, the Bureau of Reclamation may have spent as much as $70 billion on water projects for agriculture since 1902 (DeBonis 1994). Automobile use is promoted rather than discouraged through subsidies to oil and gas industries that result in lower fuel prices—which encourages consumption—and through Federal and State fuel and licensing taxes that foster road improvements, which in turn stimulate more driving, taxes, and roads. Major electric power consumers are subsidized with lower rates for greater consumption, and the Bonneville Power Administration is subsidized by taxpayers, allowing it to provide extremely inexpensive electric power to its customers. Taxpayer subsidies on public lands are estimated at $700 million for below-cost timber sales and $95 million for below-cost grazing fees and wildlife control, and the patenting of public lands at $2.50 per acre of land for mining is tantamount to a substantial Federal subsidy (DeBonis 1994). In summary, all of these subsidies provide disincentives for conserving energy and resources, and in many cases promote excessive use and consumption. Obviously, many of these subsidies provide values that benefit some or all segments of society. But it should be made clear to the public that these programs have associated environmental costs and directly or indirectly influence the ability of aquatic systems in the Pacific Northwest to produce salmonids.

In addition to the above changes, we need to reconsider fundamental policies in four areas of our culture: population, economics, ethics, and education. The first three are the cultural forces that are the root causes of environmental degradation and salmonid extirpation. Education is the method by which we begin to change our minds, and of which this document and others like it are a part. Although these forces are closely interconnected, we list and discuss them separately.

8.10.2 Population Policy

Since 1870 the population of Oregon has doubled every 35 years, and the population of the Willamette Valley has tripled since 1940. The Pacific Northwest as a whole has experienced a population growth rate of 1.3% per year over the past decade, mostly in metropolitan areas (Matske 1993). Even at this slower rate, human population will double in 54 years and quadruple in 108 years. Growth will be concentrated in the current metropolitan areas, making the region even more urbanized than it is now. The salmon and spotted owl crises have made it abundantly clear that all the commodity, ecological, and aesthetic values desired by our culture cannot be met with our current and finite resource base. Thus, it is difficult to imagine how we can protect and provide these values with a population two- to four-times larger without substantial modifications in or use of water, power, food, fiber, open space, and fish. Randers and Meadows (1973) argue that the sooner we decide appropriate population levels, the more likely we will be able to choose the set of pressures we prefer to employ in stopping or reversing population and consumption growth, rather than have nature choose them.

It is also important to realize that the Pacific Northwest supports excessive populations elsewhere through exports of forest, agricultural, energy, and fishery resources. Carrying capacity and ecological integrity in our region are diminished by high population densities elsewhere in the United States and internationally. This is not to say that we should retain the resources so that we can maximize human population in this region, because the optimum human population is less than the maximum (Hardin 1968).

8.10.3 Economic Policy

As suggested above, there is a close relationship between human population size and per capita resource consumption. Ehrlich and Holdren (1973) describe this mathematically as $I = P{\cdot}F$, where I equals total impact, P is population size, and F is per capita impact. They add that this relationship is not necessarily linear, rather the factors are interconnected. For example, higher per capita resource consumption is associated with lower population growth rates and higher population size leads to greater per capita impact, especially among the very poor and very wealthy. The important issue is that, although many people consider human overpopulation a serious global issue, fewer accept that human overconsumption is equally serious. Both population and consumption must be considered to have a dramatic effect on impact because increases in either can offset decreases in the other. For this reason we spend a little more time discussing economic issues than population issues, and in relating the two.

As Packard (1960) described, the culture of the West seems centered around population growth and overconsumption of material things. However, his reflections are not new. In the Bible, Isaiah (44:14–20) discussed the difference of using wood to satisfy basic needs (warmth, cooking) and the surplus for making an idol (infinite wants), adding that his subject's inability to see the difference prevented him from correcting his error. The classical economist Mill (1857) wrote that "if the earth must lose the pleasantness that the unlimited increase of wealth and population would extirpate from it, merely to support a larger, but not a happier or better population, I sincerely hope they will be content to be stationary, long before necessity compels them to it." He added later that "a stationary condition of capital and population implies no stationary state of human improvement. There would be as much scope as ever for all kinds of mental culture, and moral and social progress . . . and much more likelihood of its being improved" (Mill 1857). Keynes (1931), like Isaiah, contrasted absolute needs or necessities that can be satisfied with insatiable needs that make us feel superior to others.

More recently, two imminent economists with global experience have argued for steady-state economies. Schumacher (1973) felt that the aim of an economy "should be to obtain the maximum of well-being with the minimum of consumption." He also argued that "production from local resources for local needs is the most rational way of economic life because dependence on distant imports and exports is highly uneconomic." Schumacher found modern economics inadequate for evaluating the value of air, water, soil, natural living organisms, natural ecological processes, beauty, health, cleanliness, and appropriate human lifestyles. He considered GNP misleading because it does not account for the destruction and depletion of natural and cultural resources. Schumacher wrote that "when the available spiritual space is not filled by some higher motivations, then it is filled by something lower." In particular he believes that "the acquisition of wealth and materials has become the highest goal, but we must develop a life-style that accords material things as secondary. The chance of mitigating the rate of resource depletion or of bringing harmony into relationships between those in possession of wealth and power and those without them is nonexistent as long as there is no idea anywhere of enough being good and more than enough being evil" (Schumacher 1973).

Daly (1973), following Mill and Schumacher, argued that both for the good of humans and the earth, economic growth should be in services and leisure versus material goods. He described extra GNP in a rich country as satisfying relatively trivial wants because it mostly goes to the wealthier classes. The benefits of economic growth, he stated, go mainly to the rich, while the costs go mainly to the poor. He described current economic growth as being in increased output of goods per hour, meaning that the value of an hour rose in terms of goods. Daly concluded that as time becomes more expensive, fewer activities are worth the time, and time-intensive activities, like friendships, care for others, education, and meditation are sacrificed. Advertising stimulates this material consumption. Daly felt that a higher relative price of materials relative to leisure and services is needed. If raw materials are held constant and if all the ecological costs of production are included, so that costs of production increase instead of decrease, the incentive for producers to expand are eliminated (Daly 1973). He states that either producers or consumers of products must internalize the pollution and species-extirpation costs of production. Low production and consumption rates mean greater life expectancy of goods and people, less time lost to production, and less resource depletion and pollution (Daly 1973).

Increased consumption and economic growth results in decreased environmental quality because the production, use, and disposition of commodities creates environmental decay. This could be corrected somewhat if difficulties in accounting for externalities, optimizing the distribution of goods and income, and considering future generations were overcome (Barkley and Seckler 1972). Goods produced at high ecological cost could also be taxed highly (Durning 1994). Since the rate of economic growth is controlled by U.S. government officials, it can be slowed. In the long run, economic policies may have as much impact on salmonids as Federal forest plans or other more direct measures aimed at protecting Pacific Northwest ecosystems.

In various societies, economic surpluses have been used by priests that control religious sites to extract tribute, feudal lords that control land to obtain rents, and State and private capitalists that control capital to gain more capital. None of these institutions has nurtured our species to develop its greatest potentials and all have been highly destructive of our natural environment. If we hope to develop human potentials and to share the earth with anything resembling the current diversity of organisms and ecosystems present, we might be wise to listen to the economists who have long called for taking a different path than one that has historically led to ruin.

8.10.4 Ethics

The third major area in which fundamental change has been called for is in the way in which we view ourselves, others of our species, and our environment. Recently the UCS (1992) stated that "a new ethic is required—a new attitude towards discharging our responsibility for caring for ourselves and for the earth. . . . This ethic must motivate a great movement, convincing reluctant leaders and reluctant governments and reluctant peoples themselves to effect the needed changes." Although the right of property ownership holds great importance to citizens of the United States, all landowners have a responsibility to practice good stewardship to ensure that the activities in which they engage do not adversely affect resources that belong to all citizens.

Many may believe the call for a conservation ethic is new, but it began in the United States as early as the 1850's in the writings of Thoreau (Nash 1989). Thoreau writes that "What we call wildness is a civilization other than our own," and "There is no place for man-worship . . . take wider views of the universe." Both in his simple life and in his writings Thoreau treated nature as an equal and implied that it should have legal rights similar to those of other minorities oppressed by the dominant culture. Later, Marsh (1864) writes that human stewardship of nature is a moral issue, not just an economic one, and he was very concerned with consumption and waste of natural resources. John Muir's journals also included references to the "rights of animals" and "the rights of all the rest of creation" (Nash 1989). The major scientific basis for natural rights for nature lies in Darwin's (1859, 1871) works. Darwin demonstrated that the process of natural selection linked all past and present species, and challenged the idea that nature exists solely to serve humans. Darwin believed in kinship and respect for our fellow organisms, and argued that moral sympathies and ethics had survival value and thus were a product of natural selection. He wrote that ethics are the basis of animal societies and of the evolution from human families and tribes to nations and international organizations.

Most of the world's great religions and cultures included liberal views on natural rights. For example, Buddha emphasized a reverent and nonviolent attitude toward all sentient beings and especially trees, expecting observers to plant and nurture them (Schumacher 1973). The early Greeks and Romans felt that humans should respect nature, and suffer when they did not (Nash 1989). Writings of Hinduism, Judaism, and Taoism include similar guidelines and warnings on the proper way in which humans should interact with nature. Thus, more modern stirrings of conscience have a long history.

Modern thought on environmental ethics was stimulated by the development of the discipline of ecology and the writings of Schweitzer and Leopold.

Ecology quantified the interdependence between plants, animals (including humans), and their environments. Schweitzer felt that proper conduct for humans was based on a reverence for life. He wrote that an ethical person "shatters no ice crystal, tears no leaf, and crushes no insect" and that killing was done only when "absolutely necessary to enhance another life and then only with compassion" (Nash 1989). He received the 1952 Nobel Peace Prize. Leopold combined ethics and ecology, writing that "conservation based solely on economics is hopelessly lopsided because it ignores elements lacking commercial value but that are essential to healthy ecological functioning. It tends to relegate to government many functions too large, complex, and dispersed to be performed by government. An ethical obligation on the part of the private owner is the only remedy" (Leopold 1949). In perhaps his most eloquent phrase, he stated "a thing is right when it tends to preserve the integrity, stability, and beauty of the biotic community. It is wrong when it tends otherwise."

Despite a long history involving many perceptive thinkers, the extension of natural rights to nature is an incredible and revolutionary thought to many people, but it is a very compelling one to many others (Nash 1989). The same incredulity, however, met the first proposals for granting rights to human groups throughout Western history. Each of the advances listed in Table **8-3** required substantial changes in the perspectives of the dominant social group. It can be argued, however, that the greater society is improved by such changes.

8.10.5 Education

Current economic thought stresses the importance of human creativity in developing a healthy economy. Older views of natural resources and labor as the only sources of capital are being expanded to include the human intellect. With this in mind, Schumacher (1973) considered education as the most vital of all resources. However, because a purely technological education limits the ways in which we can think and our views of ourselves as a species, he felt we need a rich mix of science and humanities education throughout our lives. Apparently this position is shared by a number of educators because current pedagogical approaches integrate science and humanities instruction in growing numbers of our public schools and universities. Also, many persons at both extremes of the political spectrum decry the absence of instruction on values, ethics, and the proper role of a conscientious citizen in human society and the larger ecosystem.

Environmental education, as taught in public schools, has typically consisted of descriptions of environmental problems. This is a necessary but insufficient step. What is needed is a citizenry that is aware of 1) the status and trends in our environment, 2) the physical and social causes of both, and **3)** the individual and social changes needed to place us on a sustainable path. Most importantly, we must become willing and motivated to make those changes. Just as public, university, and adult education played critical roles in expanding civil rights to all U.S. citizens, it can play a similar role in affecting how we view our environment and our role in it. Unfortunately, we have few models of sustainable resource use from which to draw, and these are largely restricted to situations where human population density was quite low by today's standards.

We in the Pacific Northwest, blessed by some of the most diverse and least disturbed ecosystems in the conterminous United States, have a rare opportunity. We can lead the rest of the Nation along a more sustainable path, or we can follow the

Table 8-3. Development of civil and natural rights in American and Western culture. Modified from Nash 1989.

Civil or natural group	Enabling legislation or doctrine
Israelite tribesmen	Ten Commandments (2500 BP)
English lords	Magna Carta (1216)
European American men	Declaration of Independence (1776)
Livestock	Martins Act (1822)
African Americans	Emancipation Proclamation (1863)
European American women	Nineteenth Amendment (1920)
Native Americans	Indian Citizenship Act (1924)
African Americans	Civil Rights Act (1964)
Endangered plant and animal species	Endangered Species Act (1973)
Natural Ecosystems	???

heavily trodden path that has emanated from western Europe and crossed this continent in a little over 500 years. We can lead the United States to a new consciousness (just as we have with air and water quality criteria, land-use planning, and recycling), or we can hope that the same institutional and ethical approaches that led us into our current resource crises can extricate us. We can choose a lifestyle that includes salmon and the ecosystems that support them, or we can continue to extirpate them both. Because we still have wild salmon, we still have a chance to save them. Because those salmon occupy an important position in our regional culture and thinking, range throughout the region, and are affected by nearly everything we do with the lands and waters, more than any other symbol they may force us to rethink the wisdom of our current culture.

This is not a new dilemma; it is as old as civilization. "How did civilized man despoil his environment? He cut or burned the timber from the hillsides and valleys, he overgrazed and denuded the grasslands with his livestock, he killed most of the wildlife, fish, and other aquatic life, he allowed his farm topsoil to erode and clog his waterways, he wasted the easily mined minerals, then his civilization declined or he moved it elsewhere" (Dale and Carter 1955). Since we cannot move elsewhere, we can either change our minds or watch the steady degradation of our resources and quality of life. The decisions we make over the next few years will govern the world we and our progeny inherit. Education is one key way in which we can begin to include present and future citizens in the decision-making process.

8.11 Summary and Implications for Salmonids

Because the health of native salmonid populations and the condition of their habitats are inextricably linked to so many aspects of resource extraction and use, protection and long-term recovery of these fish will require fundamental changes in management practices at many levels—improving fishery management and fish stocking practices, modifying land- and water-use practices, and ultimately addressing the root cultural aspects that drive the demand for natural resources. Fundamentally, effective management systems for improving salmonids and their habitats are those that, to the greatest degree possible, minimize disruption of natural ecological processes and mimic the extent and frequency of natural disturbance. Wild native fish assemblages should be encouraged to replace non-native stocks and species. Modification of fish harvest practices (e.g., reduced harvest levels, terminal fisheries, gear and angling restrictions) can help ensure adequate recruitment of spawners and minimize other ecological effects of harvesting (e.g., size selectivity, loss of nutrient inputs to streams from carcasses). Maintenance and re-establishment of natural channels and floodplain processes can be attained through active removal of humans structures, such as dams and levees, and by returning streams to more natural flow regimes.

Channels, riparian areas, and floodplains are unlikely to recover without modifying land-use systems to some degree, both in riparian and upland areas. Regardless of the land-use type, most impacts on salmonid habitats relate to the removal of vegetation and disturbance to soils, which lead to changes in the rate of delivery of water, sediment, organic debris, and nutrients to streams. Practices that limit the areal extent, frequency, and intensity of disturbance are likely to have the least impact on salmonid habitats. Impacts of forest practices can be substantially reduce by careful layout of harvest units, roads, and skid trails. Those practices that mimic the extent and frequency of deforestation from natural disturbances, such as fire, are most likely to support the hydrological and geomorphological processes that sustain healthy channels and riparian systems. On rangelands, livestock numbers, composition, and distribution, as well as the timing and duration of grazing, can all be controlled to ensure adequate vegetation remains on site to minimize erosion and hydrologic changes. Marked improvements in riparian vegetation and instream habitat conditions can be achieved by excluding cattle from the riparian zone, or by carefully controlling the timing and intensity of grazing. Increased amounts of permanent vegetative cover are also needed on agricultural lands to keep soil and chemicals out of streams. Mined areas can be restored to natural contours and vegetative cover, and contamination of streams with mining effluents can be reduced with containment and treatment. Urban land can be restricted from expanding into relatively undisturbed systems through zoning and higher-density housing.

Salmonids are also likely to benefit from increased planning at the regional level. To an increasing degree, State and Federal resource management agencies are developing cooperative programs for salmonid conservation and restoration; this coordination of effort is essential for addressing conservation at the watershed, basin, and region levels. And finally, conservation of salmonids fundamentally comes down to the behavior and actions of individual citizens. Simpler, less consumptive lifestyles and lower reproduction rates lead to reduced demand for resources, the extraction of which directly or indirectly affects salmonid habitats. Residents of the Pacific Northwest are revealing increased concern with the survival of salmon and the goals of our civilization. Communities in the region can control their futures by establishing management systems based on a vision of desirable communities, landscape condition, and fish and wildlife populations—or they can allow continued unrestrained development at the expense of aquatic ecosystems and the salmonids they support. Already the landscape of the region has changed markedly. This trajectory can be reversed or accelerated only by the aggregate desires of its citizens. Humans are the critical variable in ecosystem management.

9 Relevant Federal Laws for Protecting and Restoring Salmonid Ecosystems on Nonfederal Lands

Three bodies of Federal law are most often cited in reference to the protection and restoration of salmonids and their habitats. The Clean Water Act (CWA) commonly refers to a group of laws intended to protect the quality and biological integrity of the Nation's waters (codified as 33 USC 1251 *et seq.* and 33 USC 1311 ***et*** *seq.* are the Act itself [CWA, PL 92-500 19721 also known as the Federal Water Pollution Control Act [FWPCA 19481 as amended by CWA [PL 92-500 1972 and PL 95-217 19771, and the Water Quality Act [WQA, PL 100-4 19871). The National Environmental Policy Act (NEPA, codified as 42 USC 4321 *et seq.,* passed as PL 91-190 1970, and its amendment the Pollution Prosecution Act [PPA, codified as 42 USC 4321, passed as PL 101-593 19901) as well as the Endangered Species Act (codified as 16 USC 1531 *et seq.*, passed as PL 93-205 1973) addresses habitats and organisms.

A fourth, less well known law, the Food Security Act (FSA, bound as 99 Stat 1354, passed as PL 99-198 1985) seeks to discourage land uses in sensitive areas (i.e., erodible soils and wetlands) through controls on Federal assistance programs. The goals and certain sections of these Federal laws that explicitly pertain to Federal and nonfederal landowners are discussed in the sections that follow. Consequently, this chapter provides only a brief overview of these laws; an exhaustive review of their effectiveness in protecting and restoring salmonids exceeds the scope of this document.

9.1 Clean Water Act (CWA)

The goals and policy of the CWA are to restore and maintain the chemical, physical, and biological integrity of the Nation's waters; to eliminate the discharge of pollutants into waters; to attain water quality that provides for the protection and propagation of fish, shellfish, and wildlife; and to develop and implement area-wide waste treatment management to control pollutant sources.

Several sections of the CWA pertain to habitat restoration. Section 2 requires States to identify areas with substantial water quality control problems (§§ 2, 31, 32, 33(a), 34 of the CWA; § 35 of FWPCA; §§ 101(d) and 101(e) of WQA [codified as 33 USC 1288])[2]. After consulting with local governments and contiguous States, States shall develop effective area-wide waste management plans for all wastes in the area. The plans shall identify and set forth controls for municipal, industrial, agricultural, silvicultural, and mining wastes to protect ground- and surface-water quality. The first sentence of Section 303 (WQA, codified as **33** USC 1311) directs States to promulgate water quality standards that protect fish and wildlife. It also obligates States to estimate total maximum daily loads for pollutants to assure protection of balanced indigenous populations of fish, shellfish, and wildlife. The WQA defines pollution as the man-induced alteration of the ecological integrity of water. This section (33 USC 1311) also addresses the illegality of pollutant discharges: "the discharge of any pollutant by any person shall be unlawful," and subject to penalties. Section 2 (CWA; § 308(d) of WQA; 33 USC 1313) directs EPA to issue guidelines for controlling pollution from agriculture, silviculture, mining, construction and hydrological modifications, and for restoring and maintaining ecological integrity in receiving waters. Section 2 (CWA; § 305 of WQA; § 52 of FWPCA; 33 USC 1315) requires States to prepare biennial reports describing the quality of all State waters; the degree to which those waters provide for the protection and propagation of balanced populations of shellfish, fish, and wildlife; and additional actions needed to achieve such objectives. Finally, Section 401 of the original 1948 law (§ 2 of CWA; §§ 61(b) and 64 of FWPCA; 33 USC 1341) directs Federal permittees involved with activities resulting in a discharge to certify that the discharge will comply with water quality standards.

'Agencies often refer to this section as "Section 208," a number derived from a version appearing as a committee print. See: U.S. Senate, 1977, Committee on Environment and Public Works, *The Clean Wafer Act and Changes Made by the 1977 Amendments,* 95th Congress, 1st Session, Committee Print 95-12, p. 34-41, Washington, DC: U.S. Government Printing Office. The Agencies also refer to Sections 303, 304, 305, etc. from this committee print which became parts constituting Section 2 of CWA (33 USC 1251 *et seq.).*

The **CWA** has been relatively effective in reducing pollution from point-source industrial and municipal discharges but has been much less effective in controlling nonpoint-source pollution, preventing cumulative effects on water quality, or protecting streams from habitat degradation unrelated to contamination (e.g., modification of hydrologic regimes, alteration of stream channels, introduction of non-native species) (Karr 1990; Doppelt et al. 1993; Karr 1995). In this regard, the goal of the CWA of maintaining biological integrity of the Nation's surface waters has not been met (Hughes and Noss 1992). The 1987 amendments to the CWA contain stronger provisions for controlling nonpoint-source pollution and place greater emphasis on protecting instream biological resources (Doppelt et al. 1993). In addition, EPA has begun a program to better protect aquatic biota. States have been directed to develop narrative biological criteria for streams, rivers, lakes, wetlands, and marine ecosystems (EPA 1990). Karr (1995) concluded that establishment of biological criteria (rather than solely chemical criteria) is essential for ensuring long-term protection of aquatic ecosystems.

9.2 National Environmental Policy *Act* (NEPA)

The National Environmental Policy Act (42 USC 4321 ***et seq.***) comprises two laws, the original NEPA (PL 91-190, passed in 1969 but dated 1970) and the Pollution Prosecution Act (PL 101-593 1990). Section 2 of NEPA (42 USC 4321) declares its purposes: to declare a national policy that encourages harmony between humans and their environment, reduces environmental damage, and improves understanding of ecological systems. NEPA recognizes the impact of human activity on the natural environment, particularly the profound influences of population growth, urbanization, industrialization, resource exploitation, and technology (§ 101; 42 USC 4331). The Federal government is responsible for coordinating Federal programs to assist each generation of the Nation to act as trustees for future generations, preserve a diverse environment and important natural aspects of our national heritage, and maximize the recycling of depletable resources. This policy section also recognizes the rights and responsibilities of each person to enjoy, preserve, and enhance the environment.

Section 102 of NEPA (42 USC 4332) requires all Federal agencies to administer laws and regulations in accord with the above policies and to give appropriate consideration to unquantified environmental amenities. For major Federal actions and legislation, all Federal agencies are obligated to provide a detailed statement on the environmental impact of the proposed action or legislation, any unavoidable adverse environmental effects, and alternatives. The agencies are also required to use ecological information in resource planning and development projects and to make information available to nonfederal institutions for restoring and maintaining environmental quality. In 1978, the Council on Environmental Quality published regulations to implement NEPA (40 CFR 4321 *et seq.* 1969, 1970). These regulations, among other things, require environmental analyses to consider cumulative effects, which are defined as "the impact on the environment which results from the incremental impact of an action when added with other past, present, and reasonably foreseeable future actions regardless of what agency (Federal or nonfederal) or persons undertakes such other actions." (Beschta et al. 1995). Judicial interpretation of the cumulative- effects language of NEPA has determined that agencies must consider impacts resulting from both Federal and private actions in determining cumulative effects within the area of a proposed action (Doppelt et al. 1993).

Because NEPA requires the analysis of cumulative effects, the Act has pushed the agencies toward watershed-level (or broader) assessment of environmental impacts (Doppelt et al. 1993), which should foster greater protection of aquatic ecosystems. However, the Act does not guarantee that environmental impacts of an action will be avoided or mitigated. It only requires that alternatives to the action be considered, that a thorough analysis of the expected environmental impacts associated with each alternative be performed, and that these impacts be disclosed to the public. Final decisions do not require that the identified impacts be avoided, thus environmental degradation can still occur. In addition, alternatives sometimes differ little from one another or are selected for political rather than ecological reasons. Finally, NEPA is usually directed toward individual projects rather than toward far-reaching policies. In particular, the Act has not been used to develop coordinated economic, ethical, or population policies that ultimately govern environmental quality.

9.3 Endangered Species Act (ESA)

Congress found that various species have become extinct in this country as a result of economic growth or development and that those same forces threaten or endanger other species. Section 2 **(16** USC 1531) of ESA states its purposes: "to provide **a** means whereby the ecosystems upon which threatened and endangered species depend may be conserved and to provide a program for the conservation of such endangered species and threatened species.. **..**" The final purpose is to honor the Nation's environmental treaties and conventions. Moreover, ESA explicitly describes two policies: all Federal agencies are to seek conservation of listed species and to cooperate with nonfederal agencies in resolving water resource issues in concert with the conservation of endangered species. Six sections are especially pertinent to the protection and restoration of salmonids and their ecosystems.

Section 3 (16 USC 1532) defines several terms in use throughout the Act. Critical habitat includes all areas occupied by the species as well as unoccupied areas essential for species conservation. A species includes subspecies and distinct population segments that interbreed when mature (e.g., salmon stocks). Take includes harassment, harm, pursuit, trapping, collecting, capture, or any attempt to do so. Harassment has been further defined to mean the intentional or negligent act or omission that significantly disrupts normal behavior patterns of the endangered or threatened species. Harm can include activities that result in significant environmental modification or degradation of the habitat of an endangered or threatened species.

Section 4 (16 USC 16 1533) describes the listing process. A petitioned species may be listed as endangered or threatened because of present or threatened modifications to its habitat or range, overexploitation, disease, predation, or other factors. Within 90 days of receiving a petition, the agency must publish whether such an action may be warranted. If so, a status review begins, and within 12 months of the petition the agency must publish its decision. Those species most likely to conflict with economic activities are prioritized for listing, which is based solely on the best scientific data available. Negative findings are subject to judicial review. If listed, the agency issues recovery plans and regulations to conserve the species and prohibit take or violations of the regulations. Recovery plans include critical habitat designations (that include consideration of the economic impact of the listing), necessary management actions, and objective measurable criteria for assessing status and trends for at least five years.

The remaining four sections involve implementation of recovery plans. Maximum cooperation with States is encouraged by Section 6 (16 USC 1535). Section 7 (16 USC 1536) obligates all Federal agencies to minimize placing listed species in further jeopardy and to perform biological assessments. Section 9 (16 USC 1538) makes it unlawful to take, possess, or violate any regulation pertaining to a listed species without a permit. Section 10 (16 USC 1̂539) outlines permit conditions. Otherwise prohibited acts are permitted for scientific purposes, to enhance species persistence, or if take is incidental to lawful activity. Issuance of such a permit requires a conservation plan that specifies likely impacts, steps and funding to mitigate such impacts, reasons alternative actions are not taken, evidence that the taking will not reduce species persistence and recovery, and responses to other requirements of the agency. The agency may also allow permits for a year or less if human subsistence or substantial economic losses are imminent. All permits are subject to public hearings.

The ESA has been one of the Nation's most powerful conservation acts. However, although the language of ESA emphasizes protection of habitats and ecosystems, the Act has not been interpreted to ensure proactive land management. Species listings generally occur after populations have substantially declined and their habitats have been greatly altered or degraded (Doppelt et al. 1993). Consequently, by the time a species is listed, options for recovery are often limited, and the costs associated with restoration may be high. In addition, ESA does not require review of projects that endanger entire ecosystems (Karr 1990). And finally, few species that have been listed under the Act have recovered sufficiently to be delisted; for example, "not a single fish warranted removal from the (American Fisheries Society) list because of successful recovery efforts" (Williams et al. 1989). Thus, while ESA has been useful in curbing losses of listed species and their habitat, the Act by itself is not likely to prevent general habitat degradation and additional species listings. Doppelt et al. (1993) provide a more thorough discussion of the benefits and shortcomings of ESA in protecting aquatic ecosystems.

9.4 Food Security Act (FSA)

A fourth law is especially relevant to private landowners. Within the Food Security Act (bound as 99 Stat 135, passed as PL 99-198 1985), Title XII—Conservation contains two subtitles that directly affect nonfederal landowners. Subtitle A—Highly Erodible Land Conservation (§§ 1211–1213) stipulates that any person who in any crop year produces a crop on highly erodible land shall be ineligible for any type of price support, payment, loan, crop insurance, or disaster payment. Exemptions to this section are granted only to persons actively applying conservation plans approved by the local soil conservation district. Subtitle B—Wetland Conservation (§§ 1221–1223) states that any person who in any crop year produces crops on converted wetlands shall be ineligible for any type of price support, payment, loan, crop insurance, disaster or payment. Landowners contract with the U.S. Department of Agriculture to implement a conservation plan, convert the land to its planned use, and agree not to conduct any harvesting, grazing, or tree planting unless allowed in the contract. This law provides incentives for farmers, ranchers, and silviculturalists to conserve riparian areas and wetlands—essential components of salmonid habits—to continue receiving Federal subsidies.

9.5 Summary and Conclusions

Three circumstances justify examining Federal laws in this technical foundation document. First, certain laws require decisions to be made based on scientific information or scientific research. Second, other laws mandate technical standards or scientific guidelines that must be met. Third, the laws outline management strategies that require scientific information or data to make decisions. Scientific knowledge can be incorporated into regulations pursuant to law or the law itself; however, if the scientific basis is not understood or is controversial, committees or commissions can serve formally to assess applicable state-of-the-science.

In addition to the body of law presented here, other Federal laws explicitly extend Federal help to nonfederal, private landowners. One example is the Forest Stewardship Act (PL 102-574 1992, codified as 16 USC 2101), an amendment to the Federal Forest and Rangeland Renewable Resources Planning Act (PL 94-588 1976, codified as 16 USC 1600 *et seq.*), which authorizes the Secretary of Agriculture to assist States and local foresters in establishing a coordinated, cooperative stewardship program for management of nonfederal forest lands and "the improvement and maintenance of fish and wildlife habitat" (16 USC 2101[b][6]).

Clean water law, NEPA, **ESA,** and **FSA** provide substantial Federal leadership and funds, including scientific information, to the States and local landowners to accomplish explicitly stated goals. These laws depend on scientific information to accomplish their objectives. Each act contains language linking human and ecological values, recognizing that ecologically healthy, biologically diverse environments provide healthy physical and economic environments for people.

10 Monitoring Aquatic Ecosystems

Despite the considerable effort and expense devoted to management of natural resources, including salmonids and their habitats, management activities rarely are accompanied by rigorous monitoring programs to determine whether plans are being implemented as designed and whether they are having the desired effects. The periodic reappraisal of management activities—where information is gathered to assess progress towards management goals and to redefine those goals if necessary—forms the backbone of the "adaptive management" philosophy. Many researchers and institutions have called for increased monitoring of the effects of land management activities at spatial scales ranging from sites and watersheds to basins and regions (Karr in press; FEMAT 1993; NRC 1992; GAO 1981, 1986).

Although there are many kinds of monitoring, two types are central to a conservation strategy for salmonids: implementation (or compliance) monitoring and assessment (or effectiveness) monitoring. Implementation monitoring involves determining if standards, guidelines, or prescriptions of a particular plan or program are being followed. Assessment monitoring is intended to evaluate whether implementation of the plan or program is achieving management objectives. A review of these two types of monitoring is instructive of the pitfalls likely to befall implementation of a salmonid conservation strategy.

10.1 Examples of Existing Implementation (Compliance) Monitoring Programs

With respect to salmonid conservation activities, the purpose of implementation monitoring is to determine if a landowner is correctly applying the standards, guidelines, or prescriptions designed to protect and restore aquatic and riparian ecosystems. Specific standards are likely to vary with the nature of the activity and the particular species of concern. Such standards might ensure maintenance of adequate riparian buffers, avoid sensitive hillslopes, conform roads to a watershed road plan, prescribe structures (e.g. fences, settling basins for sediments) be built to specifications, and verify elements of a watershed analysis. Monitoring for compliance with conditions identified in these plans could involve remote sensing of the management activity as well as site visits. Part II of this document provides a detailed discussion of elements likely to be included in a habitat conservation plan, as well as a suggested approach for compliance monitoring.

There are at least four Federally mandated, State-implemented, compliance-monitoring programs that provide insights into the development and implementation of sound monitoring for salmonid conservation practices. These are the Environmental Protection Agency's (EPA) National Pollution Discharge Elimination System (NPDES), the U.S. Army Corps of Engineer's Section 404 wetland mitigation and permitting program (WMPP), the Forest Services's (FS) Best Management Practices program (BMP), and the EPA's Rural Clean Waters Program (RCWP). Each of these programs has recently been reviewed and limitations identified.

NPDES is a permitting process for regulating emissions of pollutants from point-source facilities. The NPDES process requires monitoring pollutants characteristic of the particular type of discharge. This means that few facilities monitor the same constituents. An examination of chemistry data available from industries discharging waste into the Willamette River, Oregon, found that no quality assurance information (blanks, replicates, calibration standards, reference standards) was provided, which prevented quantitative assessment of data quality (TT 1992). Detection limits. occasionally were not reported. Other problems included inconsistent permit requirements, variables, analytical techniques, and reporting units, as well as a lack of an electronic database. Such data, whether from point or diffuse sources, have poor utility for quantitative evaluations of the degree to which dischargers treat effluent.

WMPP is an outgrowth of the Clean Water Act requiring mitigation for "unavoidable" wetland destruction by construction of a new wetland of similar size and type to ensure no net loss of wetlands. The WMPP's of several States were evaluated by a group of collaborators, namely, Gwin and Kentula (1990), Gwin et al. (1991), Holland and Kentula (1992), Kentula et al. (1992), and Sifneos et al. (1992a, 1992b). These researchers concluded that 1) objectives of created wetlands are frequently unclear; 2) project plans often indicate unfamiliarity with the literature and past failures as well as poor understanding of the surrounding landscape; 3) projects are rarely designed or constructed as required in the permits; 4) data collection and storage are generally inadequate for effectively assessing compliance (particularly for area affected, vegetation, and hydrology), and 5) implementation monitoring is insufficient. They called for staffing by professional ecologists, specifying objectives and numerical criteria, increased monitoring and verification at all project phases, and computerized databases. They added that monitoring should be sufficient to determine compliance with specific permit requirements, as well as to evaluate the effectiveness of the project in restoring wetland elements and processes (assessment monitoring). Area-wide reports on patterns and trends in implementation should be regularly produced.

The Forest Service's best management practices (BMPs) are intended to restore and protect streams by facilitating natural riparian vegetation condition and slowing sediment delivery. BMPs were evaluated for the Clearwater National Forest in Idaho by Rhodes et al. (1994), who found that the guidelines were too general to be effectively implemented and evaluated, and they often were contradictory. Moreover, guidelines were not framed within an ecosystem or watershed approach. The 1976 forest plans also lacked an ecological context as did the models from which they were generated. Unrealistic timber targets and recovery assumptions resulted. Similarly, the modeled trends for road recovery, sediment delivery, and substrate sedimentation did not agree with observed conditions either before or after the treatments. Rhodes et al. found, therefore, that the models only gave the illusion of rigor. Riparian timber harvest schedules were violated, partly because the extent of riparian areas was underestimated. Water quality standards were considered too obtuse to be usefully assessed and too permissive to protect fishery resources. Post-harvest timber recovery was assumed, but inadequately monitored because of the subjective, qualitative, and cryptic nature of the data. In addition, criteria were lacking altogether for monitoring and assessing watershed condition, fish habitat, and fish populations (assessment monitoring issues). The FEMAT (1993) and PACFISH (FS and BLM 1994a) standards and guidelines were written to correct many of these shortcomings.

The Rural Clean Water Program was a Federally cost-shared nonpoint source pollution control effort to improve water quality in 21 watersheds across the country. Projects included Tillamook Bay, Oregon, and Rock Creek, Idaho, and all involved U.S. Department of Agriculture and EPA participation together with State and local coordination and monitoring. In their evaluation of the program, Gale et al. (1993) list several lessons relevant to compliance monitoring. A thorough program evaluation by technical experts should be planned, funded, and scheduled at its initiation. Technical assistance, workshops, and periodic onsite evaluations should be included. Standardized, streamlined, annual, and final reports of activities and areas affected should be required. At the beginning, specific and measurable objectives must be set at the watershed level by representatives from the community and project agencies. These objectives should reflect desired outcomes but allow modification with increased knowledge. A computerized database, aerial photographs, and geographic information system are effective tools for tracking and reporting on project implementation, but regular visits to landowners reduce the number of misunderstandings.

The above examples suggest that a successful implementation monitoring program should consist of several key elements. Foremost, implementation monitoring programs must be adequately funded and staffed by ecologists with experience in geomorphology, hydrology, soils, vegetation ecology, fisheries ecology, database management and GIS, and geography. Specific objectives, project specifications, and tracking criteria must be included in plans, and remote sensing and periodic site visits are essential. Information will be most useful if it is entered into a computer database, using standardized streamlined forms, portable data recorders, or both. Periodic status and trend reports should be produced and the program should be technically re-evaluated every few years. The NRC (1992) concluded that many restoration projects failed because project specifications were ignored, insufficient ecological knowledge was incorporated in the planning and installation, specific objectives and criteria for tracking and redirection were lacking, and pre- and post-evaluations were omitted.

10.2 Examples of Existing Assessment (Effectiveness) Monitoring Programs

The purpose of assessment monitoring relative to salmonid conservation planning is to determine the degree to which permit compliance results in maintaining or improving habitat conditions and salmonid populations. It is essential that this connection be made because the goals of habitat conservation plans are to protect and restore salmonids. As with implementation monitoring, much can be learned from the strengths and weaknesses of other assessment monitoring programs.

Both State and Federal agencies currently conduct assessment monitoring programs for forestry, fishery, and other water resources. Many management activities, however, are not monitored at all. For example, Frissell and Nawa (1992) indicated that large numbers of artificial habitat structures have been placed in Pacific Northwest streams without any serious monitoring program to evaluate their efficacy. There are no systematic State or multistate monitoring programs in place at present; consequently, monitoring results only have site-specific or basin-level applicability at best.

The States of Idaho, Oregon, and Washington all have programs and written protocols for monitoring salmonids, salmonid habitats, water quality, and benthos (Table 10-1). Only one California protocol was located, so knowledge of that State's methods is incomplete. There is considerable comparability in parameters and methods among the States of EPA Region X (Idaho, Oregon, Washington). This is largely a result of three parallel developments. Biologists from the water quality agencies began working in 1990 to develop and test a common set of bioassessment and physical habitat protocols, culminating in publication of the biological monitoring handbook of EPA's Region X (Hayslip 1993). In addition, California, Oregon, and Washington are conducting regional EMAP (EPA's Environmental Monitoring and Assessment Program) surveys that use common field protocols. State fishery agencies in Oregon and Washington have also developed comparable modifications of Hankin and Reeves' (1988) methodology through cooperative work with the Forest Service (FS), Bureau of Land Management (BLM), and Washington's Timber, Fish and Wildlife program. In addition, Oregon has conducted since 1990 a stratified, random-sampling survey of the spawning habitat of coho salmon in coastal streams (Jacobs and Cooney 1993).

Several Federal agencies have begun assessment monitoring programs for streams in the region. The Geological Survey characterizes basins, segments, and reaches through a subjective site selection process and on a nine-year rotation. The FS and BLM inventory wadeable streams with a projected return interval of ten years, but the watershed selection is subjective. The National Biological Service (NBS) aggregates available biological data and published information to produce periodic reports. The EPA has initiated regional stream monitoring based on a statistical sampling survey design, a four-year return interval, annual sampling, and annual revisits to a subset of sites.

At the monitoring-design level, several of these programs present serious shortcomings to assessment monitoring of Federal and nonfederal lands and waters as it relates to salmonid conservation. Subjective site selection precludes use of inferential statistics and extrapolation to unsampled sites, making a regional or multistate assessment impossible. Instead, the data are only applicable to the sites selected, and they are prone to selection biases and statistical inconsistency. In addition, return intervals at the decadal scale hinder trend detection, which is a function of the rate of environmental change relative to the number of years the population is sampled. Stevens (1994), Larsen et al. (**1994,** *1995*), and references cited therein provide further explanation of these issues. Aggregations of available data, such as those of the NBS, suffer from the limitations of the sampling designs from which they were produced (mostly subjective) as well as differing indicators and sampling methods. At best these aggregations produce qualitative information on site-specific conditions; they cannot be used for quantitative estimates of status or trends in aquatic ecosystems or biota. The inventory approach favored by the FS and BLM requires walking all stream reaches, and thus provides considerable site-specific information about each one. This level of effort is extremely expensive and time consuming; it is also more likely to employ qualitative indicators that are much less sensitive to trend detection than quantitative indicators.

There is somewhat more consistency among the Federal monitoring programs at the reach scale. All three field programs use ten or more transects selected in a randomized systematic manner from which samples are taken. In addition, common indicators are used, but these differ in rigor (Table 10-2). Each program has its strengths ,and weaknesses, but greater quantification and increased completeness of indicators raises the likelihood of accurate and precise assessments of status and trends.

There is a clear need for all Federal and nonfederal institutions that are monitoring salmonid ecosystems to adopt a common sampling design, indicators, and sampling protocol. Indicators should be based on quantitative measurements to the greatest degree possible to reduce measurement variance and to provide early detection of trends. Probability-based (e.g., randomized systematic) sampling designs should be implemented to facilitate regional or basin-level population estimates, and to ensure data are collected in the most cost-efficient manner.

10.3 Sampling Design Considerations

As noted above, temporal and spatial trends in the condition of aquatic ecosystems can be most effectively assessed using a systematic, randomized sampling design. Several recent studies document the efficiency of random sampling compared with subjective sampling of aquatic systems. Landers et al. (personal communication) demonstrated that results from subjectively chosen sites differ from those obtained from randomly selected sites. They

Table 10-1. Monitoring parameters of Pacific Northwest States*

Parameters	California	Idaho	Oregon	Washington
Temperature (recording)	✓	✓	J	J
Dissolved oxygen	✓	✓	✓	J
Conductivity	J	J	✓	✓
pH	J	J	✓	✓
Statistical sampling design (sitelstation)	✓/✓	/✓	✓/✓	✓/✓
Qualitative				
Percent fines		J	✓	✓
Embeddedness	✓	✓	J	✓
Fish cover	✓	J	J	J
Velocity/depth		✓	J	J
Channel shape		J	✓	J
Pool/riffle		✓	J	J
Width/depth		✓	J	J
Bank stability		✓	✓	✓
Bank vegetation		✓	J	✓
Riparian buffer		✓	J	✓
Stream disturbance		J	J	J
Pool character		J	J	J
Winter refugia		J	J	J
Canopy cover		J	✓	✓
Stream/valley type		J	J	J
Channel sinuosity		J	J	J
Habitat units		✓	✓	✓
Quantitative				
Discharge	✓	J	J	✓
Depth and widths	✓	✓	J	✓
Gradient	✓		J	✓
Bottom substrate	J	✓	J	✓
Large wood	✓		✓	✓
Residual pool depth	J	J	J	✓
Insolation		J	J	
Canopy closure (densiometer)	✓	✓	✓	✓
Bank character	J	J	✓	✓
Benthos (quantitative)		J	J	✓
Salmonid spawning	J	J	J	✓
Fish (quantitative)	J	J	J	✓
Reference sites (regional)		J	J	✓

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* From Ralph (1990), Burton et al. (1991), Cowley (1992), Chandler et al. (1993), Clark and Maret (1993), Hayslip (1993), MSG (1993), Runyon (1994), Schuett-Hames and Pess (1994), and Schuett-Hames and Pleus (1994).

Table 10-2. Reach-level monitoring parameters of Federal Programs in the Pacific Northwest*

Parameters	Forest Service1 Bureau *of* Land Management	U.S. Environmental Protection Agency (EMAP)	U.S. Geological Survey (NAWQA)
Statistical sampling design (sitelstation)	/✓	✓/✓	
Regional reference condition		J	J
Algae (optionallcore)		✓/	/✓
Benthos (optionallcore)	✓/	/✓	/✓
Fish assemblage (optionallcore)	✓/	/✓†	/✓
Salmonid spawning (optionallcore)	✓/	✓/	
Riparian bird assemblage (optionallcore)		J/	
Microbial respiration (optional/core)		✓/	
Major cations and anions (quantitative)		✓	
Nutrients (quantitative)		✓	J
Iron and manganese (quantitative)		✓	J
Turbidity and color (quantitative)		J	✓
pH and conductivity (quantitative)		✓	✓
Dissolved oxygen (quantitative)		J	J
Temperature (quantitative)	✓	J	J
Depth and width (quantitative/qualitative)	✓	✓/	✓/
Habitat type (quantitativelqualitative)	✓	/✓	/✓
Large woody debris (quantitativelqualitative)	✓	✓/	
Fine sediment (quantitative/qualitative)		✓/	
Bank height (quantitativelqualitative)		J/	✓/
Incision (quantitativelqualitative)	J	J/	
Undercut (quantitativelqualitative)	J	J/	✓/
Gradient (sitelmap)	J	J/	
Sinuosity (quantitativelqualitative)		J/	
Aspect (quantitativelqualitative)	J	J/	
Canopy cover (quantitativelqualitative)	/✓	J/	
Substrate size (quantitativelqualitative)	/✓	/✓	/✓
Embeddedness (quantitative/qualitative)	J	/✓	/✓
Riparian vegetation structure (quantitativelqualitative)	/✓	/✓	/✓
Fish cover (quantitativelqualitative)	/✓	/✓	/✓
Human disturbance (quantitativelqualitative)	/✓	✓/	
Discharge (quantitativelqualitative)	/✓	✓/	✓/
Floodplain width (quantitativelqualitative)	/✓	/✓	✓/
Bank erosion (quantitativelqualitative)	J		/✓
Channel type (quantitativelqualitative)	/✓		/✓
Thalweg profile (quantitativelqualitative)		✓/	
Person hours/mile (region/site)	3M‡/10	0.012M§/60	3M¶/50

* From Platts et al. (1987), Cuffney et al. (1993), Dolloff et al. (1993), Gurtz (1993), Hughes (1993), Klemm and Lazorchak (1993), Meador et al. (1993a), Meador et al. (1993b), FS (1993b), and Hayslip et al. (1994).

† Includes amphibians.

‡ Assumes 300,000 stream miles in region.

§ Assumes a random sample of 400 sites.

¶ Assumes application of methods to all stream miles.

also indicate that subjectively chosen sites can be highly unrepresentative of ecoregions. Larsen (1995) showed that 200 random sites provide the same statewide and ecoregional Index of Biological Integrity (IBI) scores as 7000 subjective sites used by Ohio EPA. Paulsen et al. (personal communication) reported that random samples provide significantly different results than subjective site selection in State, multistate, and national surveys with the probability design indicating markedly greater environmental impact in all cases.

A second reason for a randomized regional sample survey is that current monitoring hinders regional evaluations of ecosystem conditions. For example, Henjum et al. (1994) found that current data collections are inconsistent and inadequately synthesized, which precludes comprehensive status-and-trend assessments. They recommended that the Federal government establish a comprehensive, quantitative biological monitoring program for the region because of the absence of a sufficient database. They also urged that the program be founded on an appropriate sampling design for tracking ecological condition and trends.

10.4 Biological Indicators

Although there has been a perception that biological indicators are both excessively costly to sample and too variable to allow detection of status and trends, recent studies have found the opposite (EPA 1987, 1990; Plafkin et al 1989; Rhodes et al. 1994). Adler (1995) described the legal basis for biological criteria and analyses, and suggested applications for them in water programs. Biological assessments are most useful for detecting the severity of aquatic life impairments and the effectiveness of management actions. Bioindicators, bioassessments, and biocriteria are essential to the assessment of salmonid ecosystems, particularly when coupled with the use of abiotic indicators, which aid in diagnosing probable causes of deterioration or improvement (Paulsen and Linthurst 1994; Mitchell 1995). Bioindicators are especially useful and reliable when based on biomonitoring of multiple assemblages and when the data are evaluated through use of multiple metrics that incorporate both assemblage elements and processes (Fausch et al. 1990; Karr 1994; Barbour et al. 1995, Yoder and Rankin 1995). In the case of habitat conservation plans, biological monitoring, especially salmonid monitoring, also represents validation monitoring—it is the essential step in determining whether proposed habitat and management changes actually produce improvements in salmonid populations. Whatever the purpose, useful bioassessments must incorporate clear objectives, effective sampling and database management, and careful data analysis and interpretation.

Currently in the United States, three States use biocriteria in regulations and 23 States are using them in water resource management; all five States supporting natural runs of Pacific salmon are using or developing biological criteria (Southerland and Stribling 1995). In a study of the value of numerical versus narrative biological criteria at 400 stream sites, Ohio EPA found that 61% attained and 9% did not attain narrative criteria, while 34% attained and 44% did not attain numerical criteria (Yoder 1991). Clearly, increased rigor in sampling designs and methods, indicators, and analytical techniques produces greater precision, accuracy, and discriminatory power (Ohio EPA 1992).

10.5 Summary

We believe that successful salmonid conservation will be directly related to the human and fiscal resources invested in a rigorous monitoring program. Examples from wetland mitigation, forest plans, point-source discharges, and rural best management plans reveal many shortcomings in existing implementation monitoring. All four programs demonstrate insufficiently funded efforts, inadequately trained staff, unclear objectives and criteria, insufficiently used remote sensing and site visits, and lack of computerized data systems. Typically, assessment monitoring programs lack statistical designs, quantitative indicators, periodic reports and reviews, and interprogram consistency as well as the shortcomings shown by the four implementation monitoring programs. If we are to conserve salmonids and their habitats, our management actions can be treated and evaluated just as experiments are, that is, with much more rigorous design and consistent data collection at a multistate scale.